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1 **Modeling the forest phosphorus nutrition in a southwestern Swedish forest site**

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10 **Abstract**

11 In this study, a phosphorus (P) module containing the biogeochemical P cycle has been
12 developed and integrated into the forest ecosystem model ForSAFE. The model was able to
13 adequately reproduce the measured soil water chemistry, tree biomass (wood and foliage), and
14 the biomass nutrient concentrations at a spruce site in southern Sweden. Both model and
15 measurements indicated that the site showed signs of P limitation at the time of the study, but the
16 model predicted that it may return to an N-limited state in the future if N deposition declines
17 strongly. It is implied by the model that at present time, the plant takes up $0.50 \text{ g P m}^{-2} \text{ y}^{-1}$, of
18 which 80% comes from mineralization and the remainder comes from net inputs, i.e. deposition
19 and weathering. The sorption/desorption equilibrium of P contributed marginally to the supply of
20 bioavailable P, but acted as a buffer, particularly during disturbances.

21 *Keywords:*

22 forest nutrition, phosphorus cycle, nitrogen cycle, ForSAFE, dynamic ecosystem model

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23 **1 Introduction**

24 Forests are among the most important ecosystems on the planet, providing and regulating
25 multiple important services such as timber production, biodiversity conservation, carbon (C)
26 sequestration, bioenergy supply and potable water supply (Nelson et al. 2011, COM 2005).
27 Nitrogen (N) is often reported to limit growth in northern forest ecosystems (Tamm 1991; Jonard
28 et al. 2015). The increase in atmospheric N deposition due to anthropogenic activities has shifted
29 forest ecosystems from being N-limited towards being N-saturated (Aber et al. 1989; Aber et al.
30 1998), causing N leaching in some forest ecosystems (Gundersen et al. 2006; Kreuzer et al.
31 2009; Yu et al. 2016). An increased nitrogen pool in the forest can affect other nutrient pools and
32 thereby forest nutrition. It can compromise the availability of base cations by depleting soil base
33 cations through N leaching, which causes acidification and eutrophication (Driscoll et al. 2003;
34 Eriksson et al. 1992; Likens et al. 1996). Another effect of increased N status is a stimulated
35 forest growth (Reich et al. 2006; Ciais et al. 2013), leading to the limitation of other nutrients
36 such as phosphorus (P) (Aber et al. 1989; Akselsson et al. 2008). A switch from N limitation to P
37 limitation over recent decades has been found in many forest studies and experimental studies in
38 Europe (Braun et al. 2010; Jonard et al. 2015; Talkner et al. 2015; Flückiger & Braun 1999) and
39 North America (Crowley et al. 2012; Tessier & Raynal 2003; Gress et al. 2007). Such a
40 transition, from N limitation to P limitation, is not commonly detected in Swedish forests
41 (Ingerslev et al. 2001; Högberg et al. 2006) due to the generally relatively low atmospheric N
42 deposition in Sweden (Simpson et al. 2011). However, in southwestern Sweden, where current
43 and historical N deposition is highest (Akselsson et al. 2010), it is suspected that P limitation
44 might already occur (Rosengren-Brinck & Nihlgård 1995; Akselsson et al. 2008).

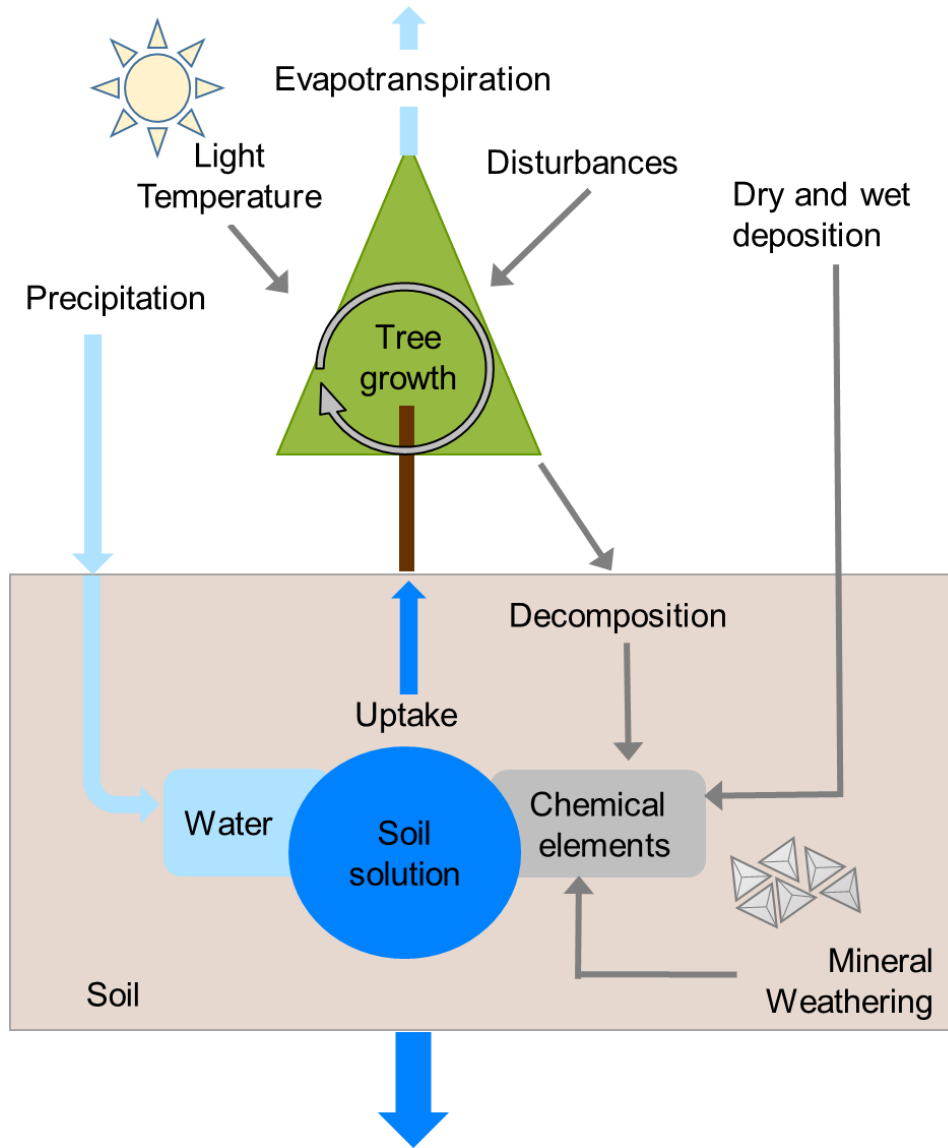
45 Forest P limitation has long been evaluated by the needle N/P ratio (Linder 1995; Rosengren-
46 Brinck & Nihlgård 1995; Mellert & Göttlein 2012; Jonard et al. 2015; Braun et al. 2010), partly
47 because foliar nutrient concentrations and ratios are well-established indicators of nutrient
48 limitation in forest trees (Mellert & Ewald 2014; Jonard et al. 2015). Particularly because P
49 nutrition in forests is more challenging to evaluate than other nutrients, due to the high
50 uncertainties in quantifying the biogeochemical P processes (Frossard et al. 2011; Shen et al.
51 2011; Fox et al. 2011; Jones & Oburger 2011), and measuring the soil P availability (Shen et al.
52 2011; Hinsinger 2001). Due to the high uncertainties in P processes measurement, the forest P
53 cycle has not been much quantitatively investigated in field studies (Yanai 1992; Yanai 1998;
54 Jonard et al. 2009), nor has it been soundly evaluated in modeling studies (Jonard et al. 2010;
55 Achat et al. 2009; Wang et al. 2010; Yang et al. 2014; Müller & Bünemann 2014). These highly
56 uncertain P processes include atmospheric deposition (Newman 1995; Tipping et al. 2014),
57 weathering (Newman 1995; Smits et al. 2012), sorption/desorption (McGechan & Lewis 2002;
58 Frossard et al. 2011), mineralization (Bünemann 2015), and rhizosphere processes (Hinsinger
59 2001; Hinsinger et al. 2011). Nevertheless, forest P cycle and its impacts on other cycles (e.g. C
60 and N) are much less investigated in modeling studies simply due to the absence of P cycle in
61 most forest/terrestrial ecosystem models (Fontes et al. 2010; Flato et al. 2013).

62 In this paper, we first implemented a P module, which contains the biogeochemical processes of
63 the full P cycle, into the integrated dynamic forest model, ForSAFE. We then tested the model at
64 a southwestern Swedish forest site, which is at high risk of P limitation. The aims of this study
65 were: 1) to evaluate the forest nutrition (N and P) at the study site, 2) to quantify the forest P
66 cycle, especially the biogeochemical P processes, from a modeling perspective.

67 **2 Methods**

68 2.1 The ForSAFE model

69 ForSAFE is a mechanistic biogeochemical model of the forest ecosystem and was designed to
70 simulate the dynamic responses of the forest ecosystem to environmental changes (Zanchi et al.
71 2014; Yu et al. 2016). The model aggregates independent processes—chemical, physical, and
72 physiological—based on empirical evidence (Belyazid 2006; Wallman et al. 2005). These
73 independent but mutually interacting processes bring together three basic material and energy
74 cycles to form a single integrated model: 1) the biological cycle, representing the processes
75 involved in tree growth; 2) the biochemical cycle, including uptake, litter decomposition, and
76 soil nutrient dynamics; and 3) the geochemical cycle, including atmospheric deposition and
77 weathering processes (Fig. 1).



78 **Figure 1.** The ForSAFE model. Climate input parameters (radiation, temperature, and precipitation) drive vegetation
79 growth. Nutrient and water availability constrain growth to the actual biomass growth and nutrient accumulation
80 (Adapted from Zanchi et al. 2014).
81

82 ForSAFE consists of four modules based on the concepts of four established models: the tree
83 growth model PnET (Aber & Federer 1992), the soil chemistry model SAFE (Alveteg 1998), the
84 decomposition model Decomp (Wallman et al. 2006; Walse et al. 1998), and the hydrology
85 model PULSE (Lindström & Gardelin 1992). The elements simulated in the soil chemistry
86 module are nitrates (NO_3^-), ammonium (NH_4^+), base cations (calcium ions [Ca^{2+}], magnesium
87 ions [Mg^{2+}], potassium ions [K^+], sodium ions (Na^+), aluminum ions (Al^{3+}), sulfates (SO_4^{2-}),
88 chloride ions (Cl^-), hydrogen ions (H^+) and dissolved organic carbon (DOC). Among these, only
89 N and base cations are treated as macronutrients for trees and are therefore modeled in the tree
90 growth and decomposition modules. The hydrology module models the soil hydrology process
91 and traces the dynamics of soil water flows and soil water contents.

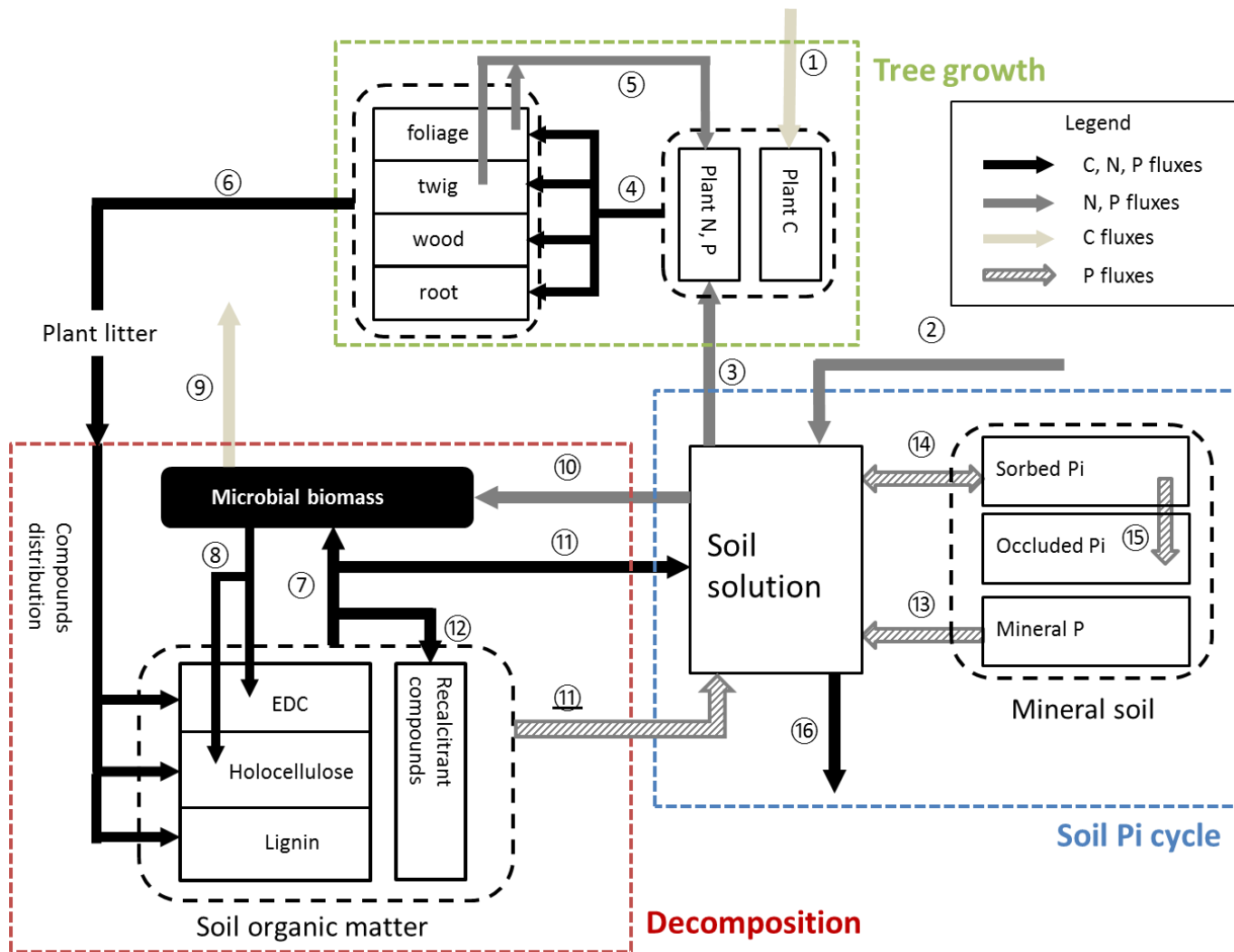
92 2.2 Inclusion of the phosphorus cycle

93 In order to better represent the biogeochemical P processes, several changes were made prior to
94 the inclusion of the P cycle. They are outlined below (Detailed description in Appendix A).

- 95 • The soil hydrology process was modified based on the new ForSAFE hydrology module
96 developed by Zanchi et al. (2016) to better represent the soil water content and soil water
97 flow, which in turn regulate the C and nutrient cycles.
- 98 • The decomposition process was complemented with concepts from three existing models
99 (Schimel & Weintraub 2003; Moorhead & Sinsabaugh 2006; Parton et al. 1988), and a
100 microbial component was explicitly added to regulate the C and nutrient fluxes in the
101 decomposition process.
- 102 • The tree growth process was modified by changes in the tree structure, plant uptake, and
103 C and nutrient allocations in order to simulate realistic plant nutrient uptake and nutrient
104 contents, particularly for those of P.
- 105 • The length of the time step was reduced from monthly to daily to better simulate the
106 hydrology process, the microbial dynamics and the P cycle, particularly the
107 sorption/desorption equilibrium.

108 The biogeochemical P processes that were taken account in the model are deposition,
109 weathering, sorption and desorption, occlusion and mineralization, which includes both
110 biological mineralization and biochemical mineralization (Fig. 2). The external inputs of P to the
111 forest ecosystem are atmospheric deposition and mineral weathering. Deposition is treated as an
112 input in the model, and weathering is simulated by the soil chemistry module. The outputs of P
113 from the forest ecosystem are through harvesting and leaching and are simulated in the same way
114 as N (Wallman et al. 2005). The P processes in the tree growth and decomposition modules are
115 also simulated in the same way as those of N, except that biochemical mineralization of P is
116 included (P mineralization catalyzed by enzymes, without releasing CO_2 , McGill & Cole 1981;
117 Oberson & Joner 2005). The P processes in the soil chemistry module are weathering,
118 sorption/desorption, and occlusion. The soil inorganic P is stored in soil solution as dissolved
119 inorganic P, in the soil matrix as sorbed inorganic P and occluded inorganic P, and in soil
120 minerals as mineral P (Fig. 2). All the inorganic P in ForSAFE is considered to be
121 orthophosphate (PO_4^{3-}), but the distinctions between different forms of orthophosphate (H_2PO_4^- ,
122 HPO_4^{2-} and PO_4^{3-}) are not simulated in the model. The dissolved inorganic P concentration is
123 determined by all the processes that exert a direct effect on it (fertilization, deposition,

124 weathering, plant uptake, immobilization/mineralization and sorption/desorption). Further details
125 of the processes descriptions can be found in Appendix A4.



126

127 **Figure 2.** Major carbon and nutrient processes in ForSAFE: ① photosynthesis, ② deposition, ③ plant nutrient uptake, ④ allocation, ⑤ retranslocation, ⑥
 128 litter fall, ⑦ microbial assimilation, ⑧ microbial decay and overflow metabolism, ⑨ microbial respiration, ⑩ immobilization, ⑪ biological mineralization and overflow metabolism mineralization, ⑫ humification, ⑬ P weathering, ⑭ P sorption/desorption, ⑮ P occlusion, ⑯
 129 nutrient leaching (percolation and surface flow). EDC: easily decomposable carbon; Pi: inorganic phosphorus.
 130

131 2.3 Site description

132 Klintaskogen is a Norway spruce (*Picea abies* Karst) forest site in southwest Sweden. The
133 average annual precipitation is 780 mm and the average annual temperature is 7.2 °C (annual
134 average between 1961 and 2010). It is a managed forest that was planted on juniperous grassland
135 in the 19th century. The latest clear-cut occurred in 1957, and the site was replanted with
136 Norway spruce. The site suffered from the wind storm *Lothar* in December 1999 and another
137 wind storm *Gudrun* in January 2005, both of which caused windthrow of the trees at the site
138 (15% and 5%, respectively).

139 The site has a very thin forest floor (3.5 cm) with 48% organic matter content. The top 50cm of
140 the mineral soil is sandy and acidic, featured with very low base saturation and high occupation
141 of Al³⁺ in the exchangeable sites. The soil is categorized as dystric podzols.

142 The measurement values of forest inventory data and soil chemistry data for the model
143 evaluation are presented in Yu et al. (2016). The soil inputs and model implementation are given
144 in the supplementary material.

145 2.4 Sensitivity analysis and model calibration

146 The parameters used in ForSAFE include parameters of tree growth, decomposition,
147 mineralization, and P sorption/desorption (Given in supplementary material). Compared to the
148 previous version of ForSAFE, the decomposition and mineralization parameters have been
149 changed as a result of changes in model concepts, such as the incorporation of processes
150 regulated by microbial activity. A regression-based sensitivity analysis was carried out to
151 estimate the impact of the modified parameters on selected outputs and to calibrate the model.
152 Twenty-two parameters regulating decomposition and tree growth were allowed to vary
153 independently and randomly within given ranges (Supplementary material, Table S.2),
154 generating a total of 1000 parameter sets. The standardized regression coefficients (SRCs)
155 (Cariboni et al. 2007; Santner et al. 2003) of each parameter on the selected outputs—the
156 average rates of N leaching, N mineralization, P weathering, P desorption, biological P
157 mineralization and biochemical P mineralization during the period 1980–2015—were calculated.

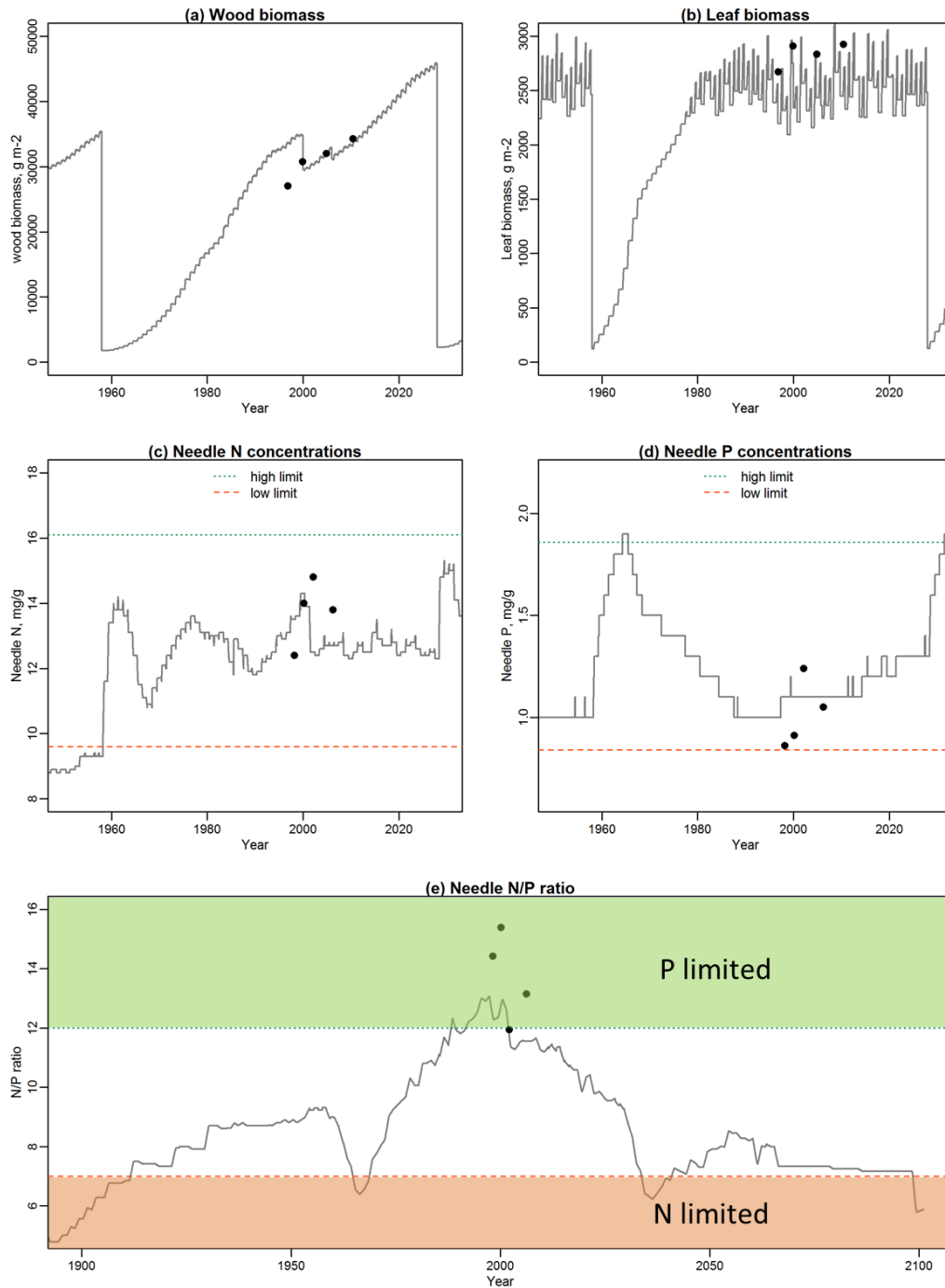
158 The performance of the model was evaluated against the measured wood biomass, needle N and
159 needle P concentrations and needle N/P ratios, the measured soil water chemistry data at a depth
160 of 50 cm, and the soil organic content (C, N, and P) of the forest floor. The best fit to measured
161 data was chosen from the 1000 parameter sets in the sensitivity analysis, firstly by comparing the
162 linear relationships between measurements and the results of the model, and secondly by visually
163 inspecting the similarities between the curves obtained from measurements and the model.

164 All statistical analysis was carried out with RStudio software (R Core Team, 2013) and the SRCs
165 were calculated using the package QuantPsync (Fletcher 2015).

166 **3 Results**

167 3.1 Model evaluation

168 The model overestimated the wood biomass by 17% and underestimated the foliage biomass by
169 13%, but the modeled trends for the change in wood biomass and leaf biomass agreed well with
170 the measurements (Fig. 3). The modeled needle N concentrations were about 5% lower than the
171 forest inventory data. Although less noticeable, the model still captured the decrease in needle N
172 concentration after 2000. The modeled needle P concentrations were within the range of the
173 forest inventory data, but the model did not capture the change in needle P concentration due to
174 storm disturbances.



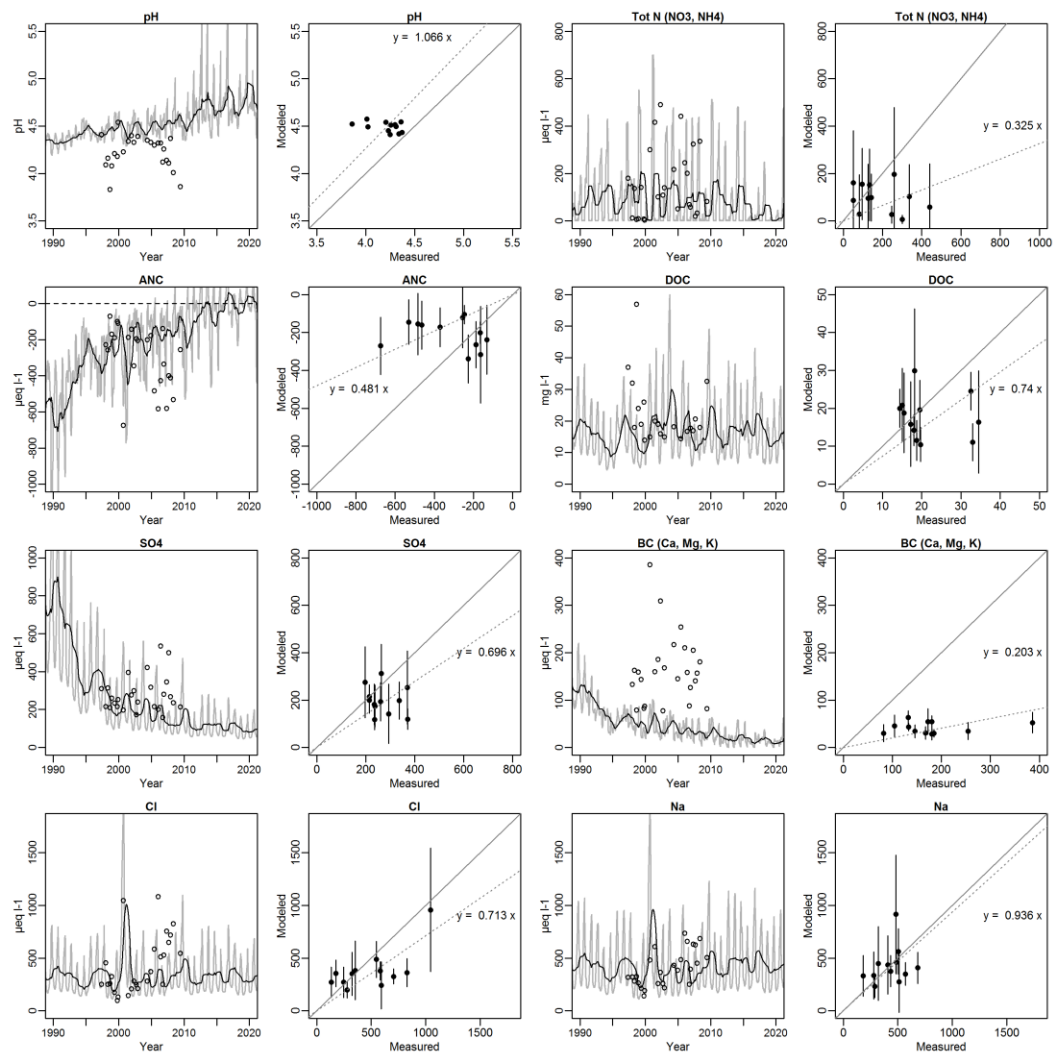
175

176 **Figure 3.** Comparison of modeled and measured wood biomass, and needle N and P concentrations during 1950–
 177 2010 (subplots (a) to (d)), and modeled and measured needle N/P ratio during 1900–2100 (subplot (e)). The black
 178 dots are calculated from measurements. Subplots (a) to (d): the low limits (dashed lines) and high limits (dotted
 179 lines) for needle N and P concentrations are based on the 5 and 95 percentile values given in ICP (International Co-
 180 operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests) report (Fischer & Lortenz
 181 2011); Subplot (e): the lines/areas of N/P ratios for N and P limitations are based on estimates by Mellert and
 182 Göttlein (2012)’s review, in which new threshold values and ratios for N and P for Norway spruce are derived from
 183 a literature compilation.

184 The modeled needle N/P ratio generally increased from 1900 until it reached a peak around
185 2000, after which it decreased (Fig. 3). Although the occurrence of the P limitation was captured
186 by the model, the needle N/P ratio was considerably underestimated around 2000, indicating that
187 forest P nutrition might be worse than the model prediction.

188 The simulated soil water chemistry generally agreed with the measurements available in 1997–
189 2009, although there was a major discrepancy in the modeling of base cations (Fig. 4). The
190 model captured the temporal trends and the ranges of the measurements for the majority of the
191 chemicals in soil water, but the modeled yearly mean values were mostly lower than the
192 measurements. We observed that N was leached out over the whole monitoring period, and N
193 leaching increased after 2000 following the storms. The model results confirmed that N was
194 leached out continuously, and the storm disturbances caused a peak in N concentration after
195 2000. Although the linear correlation between modeled pH and measured pH seems good, the
196 model actually overestimated the soil water pH by about 0.3 units and failed to capture the
197 temporal trend of pH.

198 C, N and P contents in soil organic matter (SOM) were only measured in 2010, and the model
199 predicted the SOM C and N contents very well, but clearly overestimated the SOM P content
200 (Fig. S1). As only one measured value was available, it was not possible to evaluate the
201 predicted change in SOM C, N and P contents in this study.



202
 203 **Figure 4.** Comparison of modeled and measured soil water chemistry data. The gray curves show the modeled
 204 monthly values and the black lines the moving averages (365-day periods) of these values. ANC denotes acid
 205 neutralizing capacity, ○—measurements, and ●—yearly mean values. The dotted line and its slope give an indication
 206 of the discrepancy between the modeled values and measured data. The soil water was collected at a depth of 50 cm.

207

3.2 The long-term forest N and P budgets

208 In the model, it is assumed that N enters the forest ecosystem only through deposition and that it
 209 can be stored in tree biomass, in SOM, or leach out (Table 1). Over the simulation period, 30%
 210 of the N inputs accumulated in the tree biomass, 26% was leached out, and 44% was
 211 accumulated in the SOM. The forest receives one-third of its P input from deposition and two-
 212 thirds from weathering. P can be stored in the soil matrix, tree biomass or SOM, and a small
 213 amount of it leaches out (Table 1). Over the whole simulation period, 44% of the P input
 214 accumulated in the tree biomass, 55% accumulated in the SOM and 1% was leached out. The
 215 masses of N and P given by the model are well conserved over the 300-years period (109 938
 216 time steps), with total error level of 10^{-5} g m⁻² for N and 10^{-13} g m⁻² for P. From the forest
 217 ecosystem perspective, N and P leave the forest through harvesting and leaching, thus their
 218 budgets follow ‘deposition + weathering + desorption = harvesting + leaching + accumulation in

219 trees + accumulation in SOM[†]. The model showed that Klintaskogen forest accumulated both N
 220 (160.20 g m⁻²) and P (11.80 g m⁻²) in SOM over the simulation period.

221 **Table 1.** Cumulative sinks and sources of N and P at Klintaskogen during the period 1800–2100^a

	Sources (g m ⁻²)			Sinks (g m ⁻²)			
	Deposition	Weathering	Desorption	Acc. in trees ^b	Harvest	Acc. in SOM	Leaching
N	382.09	0	0	-5.68	121.09	165.86	100.82
P	8	14.84	0.13	-0.9	10.96	12.69	0.22

222 ^aSources refers to the inputs to the forest; Sinks refers to both outputs and accumulation in the forest. Sources =
 223 Sinks in every time step and over the whole simulation period.

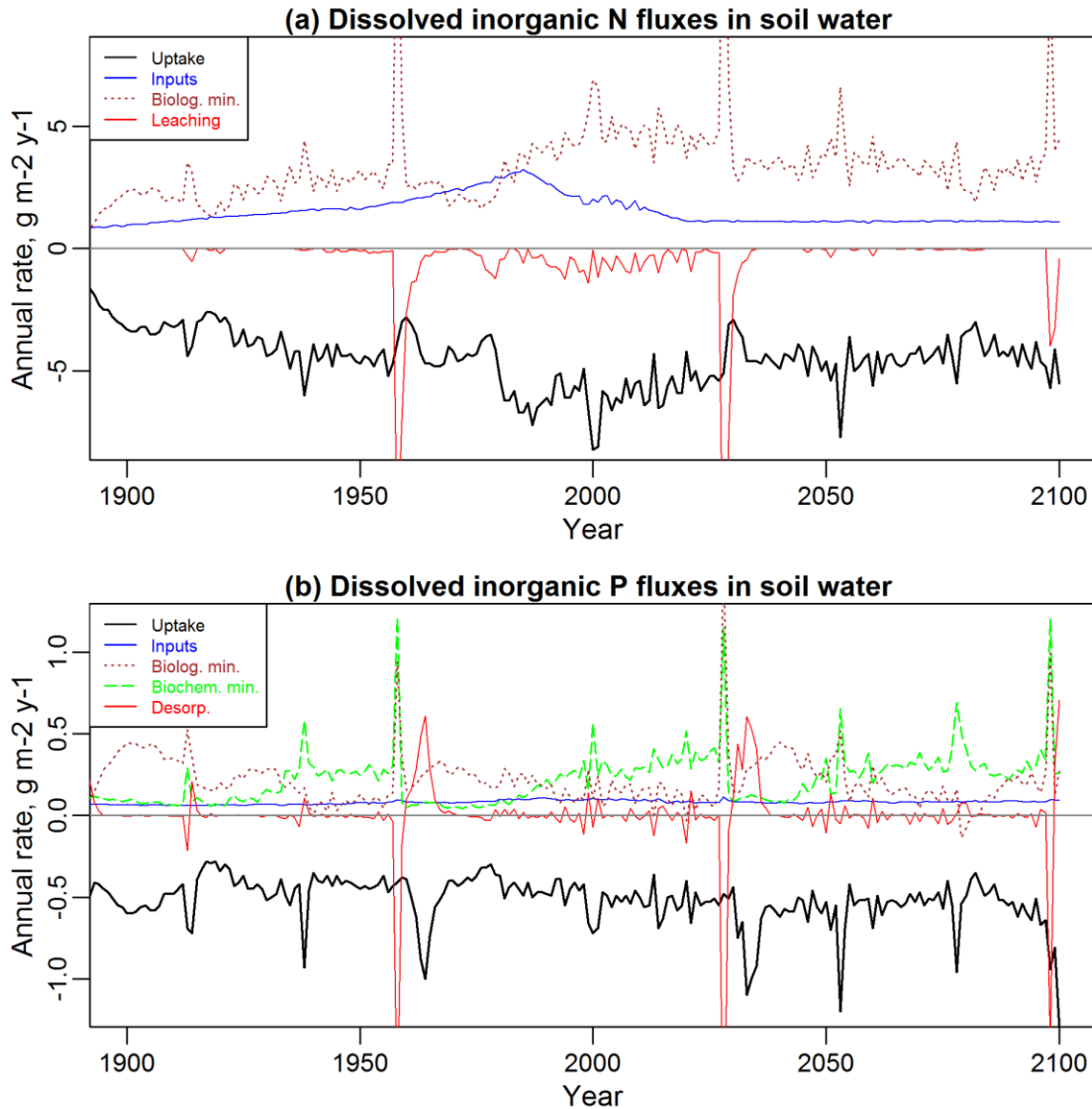
224 ^bAcc. denotes Accumulation; Acc. in tree biomass is the change in the plant pool, where negative values indicate net
 225 decreases.

226 3.3 Dissolved inorganic N and P fluxes in soil water

227 The inputs of N and P were not sufficient to support the plant uptake in the model simulation,
 228 thus the internal processes—biological mineralization, biochemical mineralization, and
 229 desorption—played important roles in supplying nutrients for uptake from the soil water (Fig. 5).
 230 Over the period of forest rotation (1900–2100), the soil water received a total of 1015 g N m⁻²
 231 (biological mineralization 707 g N m⁻²; deposition 308 g N m⁻²), of which 9% was lost through
 232 leaching, and the remainder was taken up by plants (91%, in total 922 g N m⁻²). In contrast, the P
 233 inputs (deposition and weathering) supplied only about 16% of the total influxes to soil water,
 234 while biological and biochemical processes provided 41% and 43%, respectively. Most of the
 235 dissolved inorganic P in soil water was taken up by plants (99.3%, in total 104.8 g P m⁻²).

236 As can be seen in Fig. 5(a), despite a short period of N leaching after thinning in 1912, very little
 237 N leached out during the first forest rotation (before 1957). However, after the clear-cut in 1957,
 238 N leached out continuously as a consequence of disturbances, both natural (storms) and
 239 anthropogenic (thinnings, clear cuts). In the future (2027–2097), the model predicts lower N
 240 leaching after disturbances. The N mineralization rate generally followed the pattern of plant N
 241 uptake, and periods of high fluxes were predicted after each clear-cut and thinning.

242 P inputs (deposition + weathering) did not vary significantly over the forest rotation period,
 243 whereas other fluxes (biological mineralization, biochemical mineralization and desorption)
 244 fluctuated over the whole simulation period, especially after disturbances, such as clear-cuts,
 245 thinning and storms (Fig. 5(b)). It can be seen that after each clear-cut, the release of dissolved
 246 inorganic P from the soil was increased due to biological mineralization and biochemical
 247 mineralization; most of the released P was taken up by plants or sorbed by the soil matrix.
 248 Although the desorption rate fluctuated considerably over the simulation period, the total P
 249 desorption during 1900–2100 was only -0.4 g m⁻². The total biological P mineralization was
 250 43.0 g m⁻² and the total biochemical P mineralization was 45.9 g m⁻².



251
 252 **Figure 5.** Modeled dynamics of dissolved inorganic N fluxes (a) and P fluxes (b) in soil water over the period 1900–
 253 2100. Biolog. min.: net rate of biological mineralization; Biochem. min.: rate of biochemical mineralization;
 254 Desorp.: desorption. In (a), plant N uptake + N leaching = N inputs + Biolog. N min., where N inputs include only
 255 deposition. In (b), plant P uptake = P inputs + Biolog. P min. + Biochem. P min. + Desorp. P, where P inputs include
 256 deposition and weathering. P leaching was ignored in (b) due to its low amount.

257 **3.4 Sensitivity analysis**

258 As shown in Table 2, the weathering of P is the least sensitive to changes in the selected
 259 parameters, showing less than 5% difference between the 10th percentile value and 90th percentile
 260 value. N leaching seems to be most sensitive to changes in the selected parameters, showing a
 261 difference of two orders of magnitude between the 10th percentile value and the 90th percentile
 262 value. For the other parameters studied (N mineralization, biological P mineralization,
 263 biochemical P mineralization and P desorption), the variances between the 10th percentile value
 264 and the 90th percentile value are within one order of magnitude.

265 **Table 2.** The 10th percentile values, median values and 90th percentile values of the selected outputs in the sensitivity
 266 analysis. The values are the average rates ($\text{g m}^{-2} \text{y}^{-1}$) between 1980 and 2015.

Selected outputs	10th percentile	Median	90th percentile
N leaching	0.015	0.63	1.028
N mineralization	3.09	5.14	6.10
Biolog. P min.^a	-0.059	0.26	0.56
Biochem. P min.^b	0.053	0.20	0.55
P min. Total	0.23	0.49	0.79
P weathering	0.070	0.071	0.072
P desorption	-0.0016	-0.00023	0.0066

267 ^aBiological P mineralization

268 ^bBiochemical P mineralization

269 In general, the adjusted value of R^2 of the first-order regressions between the selected parameters
 270 and outputs are very low (Supplementary material, Table S.3), indicating that the model is highly
 271 non-linear and that a single parameter can only explain a very small fraction of the variance in
 272 the output. However, the fraction of humification during DOC decomposition seems to have a
 273 much stronger impact on the selected outputs than other parameters.

274 4 Discussions

275 4.1 Forest nutrition at the study site

276 Swedish forests are mostly limited by N due to low N deposition (Akselsson et al. 2007;
277 Akselsson et al. 2008). Previous forest studies in Sweden have shown that Swedish forests
278 usually have either relatively high needle P concentrations (Vestin et al. 2013; Rothpfeffer &
279 Karlton 2007) or low needle N/P ratios due to very low needle N concentrations (Bauer et al.
280 1997; Anonymous 2003). However, low needle P concentrations or high needle N/P ratios (>10)
281 have been reported in southern Sweden, suggesting P limitation (Rosengren-Brinck & Nihlgård
282 1995; Majdi & Rosengren-Brinck 1994; Ericsson et al. 1995; Wallander & Thelin 2008) or co-
283 limitation by N and P (Rosengren-Brinck & Nihlgård 1995). The forest inventory data from the
284 present study show that Klintaskogen had very low needle P concentrations and very high needle
285 N/P ratios before the storm disturbance. Although the needle P concentrations seem to increase
286 somehow after the storms, the needle N/P ratios still remained high (Fig. 3). The observed N
287 leaching in soil water both before and after the storms also indicates that the site is approaching,
288 or is already at N saturation. Based on the above, we believe that Klintaskogen is already P-
289 limited today.

290 The model simulation confirms the P limitation at present day with the highest foliar N/P ratio
291 (Fig. 3), the highest N leaching (Fig. 4&5) and the lowest SOM C/N ratio (Fig. S1) of the entire
292 simulation period. It also indicates that Klintaskogen gradually changed from an N-limited to a
293 P-limited forest from 1900 to the present time, but will return to being N-limited again in the
294 future (Fig. 3). The simulated forest nutrition, i.e. the needle N, P concentrations, and needle N/P
295 ratios, is majorly determined by the plant uptake of N and P, which are dominated by the
296 mineralization rates. For the modeled N mineralization rate, besides a strong microbial impact
297 (Table 2), our model results show a correlation between N deposition and N mineralization (3:7).
298 In the forest N budget studies of Korhonen et al. (2013) and Kreuzer et al. (2009), similar
299 correlations were found under two different N deposition (7.4 and 45 kg N ha⁻¹ y⁻¹). This might
300 imply a universal correlation between N inputs (deposition) and N mineralization in the
301 coniferous forest ecosystem. But it needs to be interpreted carefully because the role of
302 biological N fixation is largely unknown in our model and in their studies.

303 In contrast, the rate of P inputs—deposition and weathering—is relatively stable over the
304 simulation period, as is the plant P uptake rate (Fig. 5). The biological P mineralization varies
305 considerably over the period, but the overall plant P uptake does not change to the same degree
306 (Fig. 5, Table 2). This probably indicates, firstly, that the P inputs do not have as strong
307 regulating effects on the P cycle as the N inputs have on N cycle; and secondly, the biochemical
308 P mineralization and P desorption can somehow compensate for the change in biological P
309 mineralization, thus leading to a more stable plant P uptake rate than N. Above all, the prediction
310 of the needle N/P ratio seems to be most influenced by the changes in N deposition, and we
311 predict that in the future, the forest will be N-limited under the low future N deposition
312 projection we used. However, this speculation holds a high uncertainty due to the highly
313 uncertain future N deposition (Fowler et al. 2013), and the specific site condition we modeled,
314 but we do believe future studies on other forest sites under different N deposition scenarios will
315 enlighten us about the interactions of nutrient cycles in the changing climate.

317 Over the past 35 years (1980–2015), it was estimated that trees took up $0.50 \text{ g P m}^{-2} \text{ y}^{-1}$, to
318 which biochemical P mineralization ($0.23 \text{ g P m}^{-2} \text{ y}^{-1}$) and biological P mineralization (0.17 g P
319 $\text{m}^{-2} \text{ y}^{-1}$) contributed 46% and 34% respectively. The remaining 20% consisted of deposition
320 ($0.027 \text{ g P m}^{-2} \text{ y}^{-1}$), weathering ($0.071 \text{ g P m}^{-2} \text{ y}^{-1}$) and desorption ($0.001 \text{ g P m}^{-2} \text{ y}^{-1}$). The plant
321 P uptake rate is difficult to measure *in situ* and is usually estimated by measuring the plant P
322 demand, assuming P equilibrium in the plant (Johnson et al. 2003; Yang & Post 2011). Our
323 modeled plant P uptake rate is in good agreement with the value reported by Johnson et al.
324 (2003) for temperate forests ($0.52 \pm 0.38 \text{ g P m}^{-2} \text{ y}^{-1}$), but is notably higher than the modeled P
325 uptake rate of coniferous evergreen forest by Goll et al. (2012) ($0.20 \text{ g P m}^{-2} \text{ y}^{-1}$). We believe
326 that our predicted P uptake rate is more realistic as the model also reproduced the wood biomass,
327 leaf biomass, and leaf P concentration well.

328 The model predicts that mineralization is the most important source of plant P uptake, which is
329 in agreement with the findings in many other studies (Jonard et al. 2009; Achat et al. 2010;
330 Yanai 1992; Attiwill & Adams 1993; Cross & Schlesinger 1995). Biochemical mineralization is
331 believed to be an important mechanism immobilizing organic P under P deficiency, and has been
332 widely incorporated into terrestrial ecosystem models (Wang et al. 2007; Jonard et al. 2010; Goll
333 et al. 2012; Yang et al. 2014; Runyan & D’Odorico 2012). Our model results indicate that the
334 biochemical P mineralization fluctuates over the whole simulation period, and responds strongly
335 to disturbances such as clear-cuts and storms, for two main reasons. The first is that we allow the
336 enzyme to cleave P not only from the stable organic matter but also from partially decomposed
337 lignin and holocellulose; and the second is that the microbial biomass increases dramatically
338 after disturbances, thus producing more enzymes, as has been seen *in situ* after logging
339 (Adamczyk et al. 2015). Our modeled biochemical P mineralization rates are thus generally
340 higher than those from other models, and the biological P mineralization rates are lower but are
341 still of the same magnitude comparable to estimates from other studies (e.g. Goll et al. (2012)
342 and Wang et al. (2007)). However, the model evaluation of P mineralization has always been a
343 problem since it is still not yet measurable in field scale. We believe that recent advances in *ex*
344 *situ* P mineralization measurement (Frossard et al. 2011; Bünemann 2015) could possibly
345 provide better information to constrain the model predictions in future.

346 Desorption of P is considered an important supplement to plant P uptake, especially in highly
347 weathered soils (e.g. tropical forests, Yang et al. 2014). Our results show that the P
348 sorption/desorption equilibrium plays an important role in regulating plant P uptake, especially
349 after disturbances such as clear-cuts (Fig. 5). But the overall contribution of P desorption to plant
350 P uptake is very small (Tables 1&2), indicating that in a sandy podzol soil, the soil-sorbed P pool
351 acts more as a ‘buffering’ pool rather than a source of P for plant uptake.

352 Little attention was paid to deposition and weathering in previous P cycle studies, due to the
353 model limitations or a lack of data. The present study clearly shows that deposition and
354 weathering are also important sources for plant P uptake (ca. 20%). From the whole ecosystem’s
355 perspective, the deposition and weathering are even more profound because they are the sole P
356 inputs to the ecosystem. Our predicted P weathering rate is very high compared to a previous
357 Swedish study (Akselsson et al. 2008), thus resulting in a higher accumulation rate of P in the
358 forest (Table 1). We have identified a possible overestimation of weathering rate in the model
359 and will continue to investigate its impact on the P cycle (Martin Erlandsson Lampa, personal
360 communication).

361 Recent studies on some German spruce forest sites have inferred that forest P cycle strategies
362 could evolve from ‘acquiring system’ to ‘recycling system’ as the soil P availability decreases
363 (Lang et al. 2017). The level of podsolization could be used an important indicator to know the
364 forest ecosystem’s P status—the later stage podsolization is, the more dominant by organic P
365 cycling (recycling system) (Werner et al. 2017). The high Al³⁺ exchangeable sites and low soil
366 C:P ratio (data not shown) at Klintaskogen indicate that the site is still at the early-mediate stage
367 of podsolization. This is also partly confirmed by our model results that weathering is still an
368 important source of plant uptake, and SOM is still able to accumulate organic P over the long
369 term. However, the model capacity to simulate the forest’s P cycling strategy needs to be
370 properly investigated in future.

371 4.3 Model development and sensitivity analysis

372 With the incorporation of the full P cycle, the new version of ForSAFE is able to reproduce the
373 observed soil water chemistry and forestry inventory data. One important improvement is that
374 the increased soil water N concentrations after storm disturbance can now be simulated without
375 changing the parameter values regulating the immobilization rate in the model (Yu et al. 2016).
376 We believe that the incorporation of microbial regulation on the decomposition process enables
377 the model to better represent the nutrient dynamics after disturbances. However, a major
378 discrepancy in the modeling of base cation concentrations occurred with the incorporation of
379 microbial processes. It requires future development of base cation cycle in the model.

380 We are not surprised that each parameter alone had a very small impact on the model outputs
381 (Table S.3) as the model is very complex and highly non-linear. The humification fraction during
382 DOC decomposition has a much greater impact on model outputs than other parameters. It
383 indicates that the dissolved inorganic/organic nutrients associated with DOC are essentially
384 regulating the nutrient cycle in our model, probably because the humification process is the
385 dominant process for SOM formation in the model. We also observed a systematic
386 overestimation of SOM P content in forest floor and a systematic underestimation of SOM C and
387 SOM N content in mineral soils (data not shown). It strongly implies that we should improve the
388 descriptions of biological, chemical, and physical processes in SOM, such as including
389 autotrophic microbial controls and vertical transport processes such as bioturbation and particle
390 fluxes (Ahrens et al. 2015).

391 The incorporation of P in ForSAFE naturally decreases the simulated productivity of the forest
392 and challenges the existing assumptions regarding C sequestration and allocation in the model.
393 The productivity change was not specifically investigated in this study as other changes might
394 also contribute to it. This must be further studied in the future because it may have a significant
395 impact on the C cycle and related ecosystem services, such as timber production and climate
396 regulation. Nevertheless, the incorporation of P cycle enables us to predict future forest nutrition,
397 to better evaluate the nutrient fluxes in soil water, and to study the organic C and nutrient cycles.
398 Although the evaluation of the model against empirical data is impossible for most of the P
399 processes due to the technical difficulties in measurements, the simulation of the P cycle still
400 provides valuable information on some of the less well-understood processes, such as P
401 weathering, biochemical P mineralization, and P sorption/desorption equilibrium.

402 **5 Conclusions**

403 The ForSAFE model was implemented with a P module containing the biogeochemical P cycle
404 and tested at Klintaskogen forest site. Both the forest inventory data and the model results
405 supported the suspicion that Klintaskogen is already limited by P. The model simulations showed
406 that between 1900 and 2000 the site switched from being N-limited to being P-limited, and in
407 future the site may return to N limitation, however, this prediction is heavily dependent on the
408 low future projection of N deposition. The model showed that at present period, Klintaskogen
409 forest took up $0.50 \text{ g P m}^{-2} \text{ y}^{-1}$, to which biochemical P mineralization ($0.23 \text{ g P m}^{-2} \text{ y}^{-1}$) and
410 biological P mineralization ($0.17 \text{ g P m}^{-2} \text{ y}^{-1}$) together contributed 80%, while deposition,
411 weathering, and desorption constituted the remaining 20%. The importance of P deposition and
412 weathering in forest ecosystems should be highlighted, firstly due to their contributions to plant
413 uptake, and secondly due to the role as the sole inputs to the system. At Klintaskogen, the P
414 sorption/desorption equilibrium contributes very little to plant P uptake in the long term, but it
415 can regulate the plant P uptake rate, especially after disturbances.

416 With the current model structure and processes, ForSAFE adequately reproduced the measured
417 soil water chemistry, the measured tree biomass and its chemical composition, including the
418 needle N and P concentrations of the study site. Although we are aware of different sources of
419 uncertainties and the study/model limitations, we have moderate confidence in the modeled
420 forest nutrition and forest P cycle. Some of the less well-understood processes that we addressed
421 here, such as P weathering, biochemical P mineralization, and P sorption/desorption equilibrium,
422 should be further investigated in both field and modeling studies.

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428

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Notations	Definitions	Location
<i>ActEvap</i> and <i>PotEvap</i>	actual and potential evapotranspiration rates, $\text{m}^3 \text{ water m}^{-2} \text{ soil d}^{-1}$	Eq. A1.1
<i>Perco</i>	percolation rate, $\text{m}^3 \text{ water m}^{-2} \text{ soil d}^{-1}$	Eq. A1.2
<i>Bypass</i>	bypass flow rate, $\text{m}^3 \text{ water m}^{-2} \text{ soil d}^{-1}$	Eq. A1.3
<i>Surface</i>	surface flow rate, $\text{m}^3 \text{ water m}^{-2} \text{ soil d}^{-1}$	Eq. A1.4
<i>moist</i>	soil moisture, $\text{m}^3 \text{ water m}^{-3} \text{ soil}$	Eq. A1.5
<i>wp</i>	wilting point of the soil, $\text{m}^3 \text{ water m}^{-3} \text{ soil}$	Eq. A1.1
<i>lp</i>	limit point for evapotranspiration of the soil, $\text{m}^3 \text{ water m}^{-3} \text{ soil}$	Eq. A1.1
<i>fc</i>	field capacity of the soil, $\text{m}^3 \text{ water m}^{-3} \text{ soil}$	Eq. A1.2
<i>fs</i>	field saturation of the soil, $\text{m}^3 \text{ water m}^{-3} \text{ soil}$	Eq. A1.3
<i>Kh</i>	unsaturated conductivity, m water d^{-1}	Eq. A1.3
<i>A</i>	area of the soil column, m^2	Eq. A1.3
<i>prec</i>	precipitation, m	Eq. A1.4
<i>z</i>	depth of the soil layer, m	Eq. A1.4
<i>t</i>	time step, d	Eq. A1.5
<i>C_{loss}</i>	mass loss rates of the decomposable compounds, $\text{g C m}^{-3} \text{ d}^{-1}$	Eq. A2.1
<i>k_{pot}</i>	potential mass loss rate constant at assumed optimal conditions, $\text{g C m}^{-3} \text{ d}^{-1}$	Eq. A2.1
<i>M</i>	mass of organic C of the decomposable compounds, g C	Eq. A2.1
<i>θ</i>	relative soil moisture, $\text{m}^3 \text{ water m}^{-3} \text{ water}$	Eq. A2.2
<i>T</i>	soil temperature, K	Eq. A2.3
<i>pH</i>	soil water pH	Eq. A2.5
<i>E_a</i>	activation energy, J mol^{-1}	Eq. A2.3
<i>R</i>	gas constant, $\text{J mol}^{-1} \text{ K}^{-1}$	Eq. A2.3
<i>T_r</i>	reference temperature, K	Eq. A2.3
<i>K_w</i>	empirical coefficient of moisture responding factor	Eq. A2.4
<i>n_w</i>	empirical exponent of moisture responding factor	Eq. A2.4
<i>K_{pH}</i>	response coefficient of acidity responding factor	Eq. A2.5
<i>m</i>	empirical exponent of acidity responding factor	Eq. A2.5
<i>DEC</i>	decomposition rate, $\text{g C m}^{-3} \text{ d}^{-1}$	Eq. A2.6
<i>η</i>	fraction of microbial assimilation in decomposition	Eq. A2.6
<i>k_{hum}</i>	fraction of stabilization (humification) in decomposition	Eq. A2.6
<i>k_{lch}</i>	fraction of DOC loss in decomposition	Eq. A2.6
<i>r_{min}</i>	mineralization rate, $\text{mg N(P) m}^{-3} \text{ d}^{-1}$	Eq. A2.7
<i>r_{blmin}</i>	biological mineralization rate, $\text{mg N(P) m}^{-3} \text{ d}^{-1}$	Eq. A2.8

r_{bcmin}	biochemical mineralization rate, mg P m ⁻³ d ⁻¹	Eq. A2.7
r_{OMmin}	mineralization rate, mg $N(P)$ m ⁻³ d ⁻¹	Eq. A2.7
CPr	C: P ratios of the decomposable compounds	Eq. A2.8
CPr_{mic}	C: P ratios of the microbial biomass	Eq. A2.8
$MaxP_{bcmin}$	maximum potential P biochemical mineralization rate, mg P m ⁻³ d ⁻¹	Eq. A2.9
Mic	microbial biomass, g C m ⁻³	Eq. A2.9
P_o	sum of organic P in <i>holo</i> , <i>lig</i> and <i>recal</i> , mg P m ⁻³ <i>soil</i>	Eq. A2.9
Cp	concentration of the dissolved inorganic phosphorous, mg P dm ⁻³ <i>water</i>	Eq. A2.9
k_{enz}	maximum rate of enzymatic mineralization, m ³ g ⁻¹ C d ⁻¹	Eq. A2.9
P_{und}	unfulfilled P demand, mg P m ⁻³ d ⁻¹	Eq. A2.10
Mic_{OM}	microbial overflow metabolism C loss, g C d ⁻¹	Eq. A2.11
Mic_{red}	microbial growth reduction due to nutrient limitation, g C d ⁻¹	Eq. A2.11
Mic_{ass}	microbial assimilation, g C d ⁻¹	Eq. A2.12
Mic_{resp}	microbial respiration, g C d ⁻¹	Eq. A2.13
Mic_{decay}	microbial decay, g C d ⁻¹	Eq. A2.14
f_{resp}	respiration rate of microbial biomass, d ⁻¹	Eq. A2.13
f_{micdec}	decay rate coefficient of microbial biomass, d ⁻¹	Eq. A2.14
r_{des}	desorption rate of P, mg P m ⁻³ d ⁻¹	Eq. A4.1
Pr_{tot0} and Pr_{tot}	total sorbed P in the soil matrix before and after the time step, mg P kg ⁻¹ <i>soil</i>	Eq. A4.1
Pr_{1d}	sorption capacity of the soil layer within one day, mg P kg ⁻¹ <i>soil</i>	Eq. A4.1
P_{bal}	total change rate of certain processes shown in Eq. B4.2, mg P m ⁻³ d ⁻¹	Eq. A4.2
r_{upt}	actual plant uptake of P, mg P m ⁻³ d ⁻¹	Eq. A4.2
r_{wea}	mineral weathering rate of P, mg P m ⁻³ d ⁻¹	Eq. A4.2
r_{dep} and r_{fert}	P deposition rate and fertilization rate, mg P m ⁻³ d ⁻¹	Eq. A4.2
v , w and x	fitted parameters from the isotopic dilution sorption experiments	Eq. A4.3, 4
T_{1yr} and T_{1d}	time in minutes for one year and one day	Eq. A4.3, 4
Cp_{eq}	equilibrium Cp at the soil solution and soil matrix interface, mg P dm ⁻³ <i>water</i>	Eq. A4.3, 4
$Clay$	soil particles smaller than 2 μm, g kg ⁻¹	Eq. A4.5
$Silt$	soil particles between 2 μm and 50 μm, g kg ⁻¹	Eq. A4.6
$Sand$	soil particles between 50 μm and 2000 μm, g kg ⁻¹	Eq. A4.5
P_{tot}	total P in the soil layer, g P kg ⁻¹ <i>soil</i>	Eq. A4.6
N_{org}	organic N in the soil layer, g N kg ⁻¹ <i>soil</i>	Eq. A4.7
$MaxPr_{tot}$	maximum soil P sorption capacity, mg P m ⁻³	Eq. A4.8
$(Al+Fe)_{ox}$	oxalate-extractable Al and Fe, mmol kg ⁻¹	Eq. A4.8
mw_P	molecular weight of P, mg mmol ⁻¹	Eq. A4.8

Q_{in} and Q	discharges of inflow and outflow of the soil layer, $\text{dm}^3 \text{ water m}^{-3} \text{ soil d}^{-1}$	Eq. A4.9
Cp_{in}	Cp in the inflow, $\text{mg P dm}^{-3} \text{ water}$	Eq. A4.9

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A1 Soil hydrology

636 The simulations of soil water contents and flows are based on the soil hydrology module of the *ForSAFE* model
637 (Zanchi et al. 2016). The soil is represented by a soil column with several layers denoting the soil horizons. The
638 water flows by percolation or surface flow. The percolation is determined by the hydraulic conductivity and
639 constrained by the capacity to receive water in the next layer, but in the last layer, it is restricted by the base flow
640 rate, assuming that the water cannot drain freely. Surface flow occurs above the uppermost layer if it becomes
641 oversaturated due to high precipitation or if percolation stops when the layers below are saturated.

$$642 \quad ActEvap = \min(1, \max(0, \frac{moist-wp}{lp-wp})) \times PotEvap \quad (A1.1)$$

$$643 \quad Perco = (\min(\min(\max(0, (moist - fc) \times z + Vol_{in} - ActEvap), Kh \times A \times t), \max(0, (fs_{next} - moist_{next}) \times z_{next}))) / t \quad (A1.2)$$

$$645 \quad Bypass = (\min((moist - fc) \times z + Water_{in} - ActEvap, \max(0, (fs_{next} - moist_{next}) \times z_{next} + Kh_{next} \times A \times t))) / t \quad (A1.3)$$

$$647 \quad Surface = (\max(0, moist_1 \times z_1 + prec_{in} - Perco_1 - Evap_1 - Bypass_1)) / t \quad (A1.4)$$

$$648 \quad \theta = \theta_0 + Vol_{in} - ActEvap - (Perco - Bypass - Surface) \times t \quad (A1.5)$$

649 where *ActEvap* and *PotEvap* are the actual and potential evapotranspiration rates ($\text{m}^3 \text{ water m}^{-2} \text{ soil d}^{-1}$); *Perco*,
650 *Bypass* and *Surface* are percolation, bypass flow and surface flow, respectively ($\text{m}^3 \text{ water m}^{-2} \text{ soil d}^{-1}$); θ is the soil
651 moisture ($\text{m}^3 \text{ water m}^{-3} \text{ soil}$); *wp*, *lp*, *fc* and *fs* are the wilting point, limit point for evapotranspiration, field capacity
652 and field saturation point of the soil, all in the unit $\text{m}^3 \text{ water m}^{-3} \text{ soil}$; *Vol_{in}* is the inflow water ($\text{m}^3 \text{ water}$), including
653 percolation and bypass flow; *Kh* is the unsaturated conductivity (m water d^{-1}); *prec* is the precipitation (m),
654 including melting snow; *z* is the depth of the soil layer (m); *A* is the area of the soil column (m^2); and *t* is the time
655 step (d). The denotation *next* indicates the next layer; *1* indicates the first layer; and *0* indicates the value from the
656 last time step.

657 The hydraulic properties (*wp*, *lp*, *fc*, *fs* and *Kh*) are estimated based on the soil input data, following the algorithms
658 used in the work of Zanchi et al. (2016).

659 A2 Decomposition

660 In *ForSAFE*, SOM refers to the solid humified compounds in the soil and contains organic C, N and P. Following
661 the concept of the *DECOMP* model (Walse et al. 1998), SOM is divided into four categories of decomposable
662 compounds: *easily decomposable compounds (EDC)*, *holocellulose (holo)*, *lignin (lig)* and *recalcitrant compounds*
663 (*recal*). SOM is fed by fresh plant litter and newly decayed microbial biomass (Figure 2- Decomposition). It is
664 assumed that plant litter contains *EDC*, *holo* and *lig*, whereas microbial necromass contains only *EDC* and *holo*. The
665 description and distribution of compounds are given in SI.

666 The carbon loss rate of the decomposable compound is calculated as:

$$667 \quad C_{loss_i} = k_{pot_i} M_i f_i(T) g_i(\theta) \phi_i(pH) \quad (A2.1)$$

$$668 \quad \theta = moist / fs \quad (A2.2)$$

669 where C_{loss_i} is the carbon loss rate ($\text{g carbon m}^{-3} \text{ d}^{-1}$), *i* denotes the compound index (*EDC*, *holo*, *lig* or *recal*), k_{pot} is
670 the potential rate constant at assumed optimal conditions ($\text{g carbon m}^{-3} \text{ d}^{-1}$), *M* is the mass of organic C of the
671 decomposable compound (g carbon), *T* is the temperature of the soil layer (K), θ is the relative soil moisture, and *pH*
672 is the soil water pH. The rate regulating functions are adopted from Walse et al. (1998),

$$673 \quad f(T) = \exp\left(\frac{E_a}{R \cdot T_r} - \frac{E_a}{R \cdot T}\right) \quad (A2.3)$$

$$674 \quad g(\theta) = \frac{K_w \cdot \theta^{n_w}}{1 + K_w \cdot \theta^{n_w}} \quad (A2.4)$$

$$\phi(pH) = \frac{1}{1 + K_{pH} \cdot [H^+]^m} \quad (\text{A2.5})$$

where E_a is the activation energy (J mol^{-1}), R is the gas constant ($\text{J mol}^{-1} \text{K}^{-1}$), T_r is the reference temperature (K), K_w is the empirical coefficient, n_w is the empirical exponent, K_{pH} is the response coefficient, and m is the empirical exponent. Values of the parameters are given in *ForSAFE parameterization*.

The decomposition rate of the decomposable compounds (total carbon loss of the compound category, DEC , $\text{g C m}^{-3} \text{d}^{-1}$) is calculated based on the mass loss rate:

$$DEC_i = \frac{C_{loss_i}}{(1-\eta_i)(1-k_{hum_i}-k_{lch_i})} \quad (\text{A2.6})$$

where i denotes the compound category; η is the assimilation factor that represents the fraction of DEC assimilated by microbes for growth; k_{hum} is the humification coefficient, which represents the fraction of DEC that forms recalcitrant compounds; and k_{lch} is the leaching coefficient, which represents the fraction of DEC that forms dissolved organic carbon (DOC) (Berg & McLaugherty 2008).

Soil organic nutrients are associated with the decomposition of C and therefore are humified to form recalcitrant compounds, enter the soil solution in dissolved inorganic forms (mineralization), or are assimilated by microbes (immobilization) (Figure 2- Decomposition). The mineralization rate (r_{min} , $\text{mg m}^{-3} \text{d}^{-1}$) can be divided into biological mineralization (r_{blmin} , $\text{mg m}^{-3} \text{d}^{-1}$), biochemical mineralization (r_{bcmin} , $\text{mg m}^{-3} \text{d}^{-1}$, only P) and overflow metabolism mineralization (r_{OMmin} , $\text{mg m}^{-3} \text{d}^{-1}$):

$$r_{min} = r_{blmin} + r_{bcmin} + r_{OMmin} \quad (\text{A2.7})$$

$$r_{blmin} = \sum_{i=1}^n DEC_i \left(\frac{1-k_{hum_i}}{CPr_i} - \frac{(1-k_{hum_i}-k_{lch_i})\eta_i}{CPr_{mic}} \right) \quad (\text{A2.8})$$

where CPr and CPr_{mic} are the C: P ratios of the compounds and microbial biomass, respectively; for N CNr and CNr_{mic} are used in Eq. B2.8.

Apart from biological mineralization, the nutrients in organic forms can also become plant available through C overflow metabolism and biochemical mineralization. Carbon overflow metabolism is the elimination of C and nutrients from microbial biomass when C is in excess (Schimel & Weintraub 2003; Manzoni & Porporato 2009). It is treated in the same way as microbial decay, except that all the overflowed C is respired. Biochemical mineralization (r_{bcmin} , $\text{mg m}^{-3} \text{d}^{-1}$, mineralization catalyzed by enzymes and without releasing CO_2 , McGill & Cole 1981; Oberson & Joner 2005) accounts only for P, and the rate is as follows:

$$MaxP_{bcmin} = \begin{cases} Mic \cdot P_o \cdot k_{enz} \cdot f(T, \theta, pH) & \text{if } Cp \leq 0.005 \\ Mic \cdot P_o \cdot k_{enz} \cdot \frac{Cp^{-0.5}}{0.005^{-0.5}} \cdot f(T, \theta, pH) & \text{if } Cp \geq 0.005 \end{cases} \quad (\text{A2.9})$$

$$r_{bcmin} = \min(MaxP_{bcmin}, P_{und}) \quad (\text{A2.10})$$

where P_{org} is the sum of organic P in *holo*, *lig* and *recl* (mg P m^{-3}); $MaxP_{bcmin}$ is the maximum potential P biochemical mineralization rate ($\text{mg P m}^{-3} \text{d}^{-1}$); Cp is the concentration of dissolved inorganic phosphorous ($\text{mg P dm}^{-3} \text{water}$); k_{enz} is the maximum rate of enzymatic mineralization ($\text{m}^3 \text{g}^{-1} \text{C d}^{-1}$); $f(T, \theta, pH)$ is the rate response function given in Eq. B.3, B.4 and B.5; and P_{und} is the unfulfilled P demand ($\text{mg P m}^{-3} \text{d}^{-1}$). It is assumed that the maximum biochemical mineralization rate is higher when Cp is low, and it has a maximum rate when the threshold concentration for plant uptake ($0.005 \text{ mg P dm}^{-3} \text{water}$, Stroia et al. 2007) is reached.

Microbes are heterotrophic decomposers that form biomass from decomposed C and nutrients. Microbes are the sole decomposer in the model. The following assumptions are made: first, the carbon-to-nutrient ratios of new microbial biomass are strictly restricted within certain ranges (Cleveland & Liptzin 2007; Parton et al. 1988; Manzoni et al. 2010); second, the microbial biomass decays at a certain rate (f_{micdec}), and the decayed microbial biomass (microbial necromass) is assumed to be *EDC* and *holo* and will be decomposed again. The change in microbial biomass is described as:

$$\frac{dMic}{dt} = Mic_{ass} - Mic_{resp} - Mic_{OM} - Mic_{red} - Mic_{decay} \quad (\text{A2.11})$$

$$Mic_{ass} = \sum_{i=1}^n DEC_i (1 - k_{hum_i} - k_{lch_i}) \eta_i \quad (\text{A2.12})$$

$$Mic_{resp} = Mic \cdot f_{micresp} \quad (\text{A2.13})$$

$$Mic_{decay} = Mic \cdot f_{micdec} \quad (\text{A2.14})$$

where Mic_{ass} is the microbial assimilation (g C d^{-1}); Mic_{resp} is the microbial respiration (g C d^{-1}); Mic_{OM} is the

720 microbial overflow metabolism C loss (g C d^{-1}); Mic_{red} is the microbial growth reduction due to nutrient limitation
 721 (g C d^{-1}); Mic_{decay} is the microbial decay (g C d^{-1}); $f_{micresp}$ is the maintenance respiration rate of microbial biomass (d^{-1})
 722 1); and f_{micdec} is the decay rate coefficient of microbial biomass (d^{-1}). The Mic_{OM} and Mic_{red} occur at nutrient
 723 deficiency, and they are dependent on the amount of nutrient deficient and the carbon-to-nutrient ratios of microbial
 724 biomass.

725 **A3 Tree growth**

726 Two main changes have been introduced to the tree structure: first, P has been added as a macro nutrient in plants,
 727 and second, a new compartment for twigs is distinguished from wood. Changes have also been made regarding plant
 728 uptake and nutrient allocation. The potential plant uptake is calculated as the growth rate of each tree compartment
 729 times the optimal nutrient content of the tree compartment during growing season of foliage and wood, and it is
 730 calculated as 110% of the root nutrient demand in non-growing season. The timing of nutrient uptake for foliage
 731 growth has been changed from only once per year at bud burst to multiple times synchronous with foliage growth.
 732 The wood growth has been changed from being limited only by plant C pool to being limited by both the C and
 733 nutrient pools in plant.

734 Four mechanisms have been introduced to regulate the N and P uptake of plants and microbes, namely plant uptake
 735 downscaling, microbial immobilization downscaling, microbial overflow metabolism and biochemical
 736 mineralization. When the nutrient demands (plant uptake and microbial immobilization) exceed nutrient availability,
 737 plant uptake and growth will first be decreased, followed by a reduction in microbial growth to reduce
 738 immobilization, and finally a reduction in microbial biomass by eliminating some carbon and nutrients. Biochemical
 739 mineralization accounts only for P and is negatively related to Cp .

740 **A4 Soil inorganic P cycle**

741 In ForSAFE, the P deposition (r_{dep} , $\text{mg P m}^{-3} \text{d}^{-1}$) is treated as model input and the weathering of P (r_{wea} , mg P m^{-3}
 742 d^{-1}) is simulated by the soil chemistry module, SAFE (Belyazid 2006).

743 The sorption/desorption process of P refers to all the inorganic P ion exchange processes (except weathering)
 744 between the soil solution and soil matrix. The desorption rate (P ions moving from the soil matrix to soil solution,
 745 r_{des} , $\text{mg P m}^{-3} \text{d}^{-1}$) is modeled as the change of total sorbed Pi in the soil matrix between two time steps (Stroia et al.
 746 2007; Messiga et al. 2012; Morel et al. 2014). It is then constrained by the sorption capacity of the soil layer within
 747 the time step (Pr_{1d} , $\text{mg P m}^{-3} \text{soil}$):

$$748 \quad r_{des} = \begin{cases} Pr_{tot0} - Pr_{tot}, & -Pr_{1d} < P_{bal} < Pr_{1d} & (a) \\ -Pr_{1d}, & P_{bal} < -Pr_{1d} < 0 & (b) \\ Pr_{1d}, & P_{bal} \geq Pr_{1d} \geq 0 & (c) \end{cases} \quad (A4.1)$$

749 where Pr_{tot0} and Pr_{tot} are the total sorbed P in the soil matrix ($\text{mg P kg}^{-1} \text{soil}$) before and after the time step, and P_{bal}
 750 is the total change rate of all the other processes ($\text{mg P m}^{-3} \text{d}^{-1}$) that exert a direct impact on Cp (fertilization,
 751 deposition, weathering, plant uptake and immobilization/mineralization) and is given by

$$752 \quad P_{bal} = r_{fert} + r_{dep} + r_{wea} + r_{min} + r_{upt} \quad (A4.2)$$

753 when P_{bal} does not exceed the range of Pr_{1d} , sorption equilibrium is reached (condition B4.1.a); otherwise maximum
 754 net sorption (condition B4.1.b) or maximum net desorption (condition B4.1.c) occurs.

755 According to Forssard & Sinaj (1997) and Fardeau (1996), the total desorbed P is calculated using the Freundlich
 756 kinetic equation, assuming that the exchangeable Pi between soil matrix and soil solution within a year is the total
 757 sorbed P in the soil matrix (Eq. B4.3). The sorption capacity of a time step (day) is defined as the exchangeable Pi
 758 within the time step (Eq. B4.4):

$$759 \quad Pr_{tot} = vCp_{eq}^w (T_{1yr})^x \quad (A4.3)$$

$$760 \quad Pr_{1d} = vCp_{eq}^w (T_{1d})^x \quad (A4.4)$$

761 where v , w and x are fitted parameters from the isotopic dilution sorption experiments, with their estimation given in
 762 ForSAFE parameterization; T_{1yr} is the time in minutes and, for Eq. D.3, equals $24*60*365$; T_{1d} is the time in
 763 minutes and, for Eq. D.4, equals $24*60$; and Cp_{eq} is the equilibrium Cp at the soil solution and soil matrix interface
 764 ($\text{mg P dm}^{-3} \text{water}$). Note here that the total sorbed Pi and daily sorption capacity are related to the equilibrium Cp .

765 The parameters (v , w , and x) in the Freundlich kinetic equation are determined through isotopic dilution experiments
 766 (Stroia et al. 2007). *ForSAFE* also offers pedotransfer functions (Eq. B4.5 – B4.7) to estimate the parameters based
 767 on the physical-chemical properties of the soils from selected studies (Stroia et al. 2007; Achat et al. 2010; Morel et
 768 al. 2014; Stroia et al. 2011; Messiga et al. 2012).

$$769 \quad v = 74.84 \times Clay + 9.40 \times Sand - 0.099 \times Fe_{ox} + 0.032 \times Al_{ox} + 1.04 \times pH + 1.12 \times P_{tot} - 14.29$$

770 (A4.5)

$$771 \quad w = 0.34 \times Silt - 1.089 \times Clay - 0.045 \times pH - 0.11 \times P_{tot} + 0.99$$

772 (A4.6)

$$773 \quad x = 0.11 \times Clay + 0.023 \times N_{org} + 0.26$$

774 (A4.7)

773 where *Clay* ($g\ kg^{-1}$) is soil particles smaller than 2 μm ; *Silt* ($g\ kg^{-1}$) is soil particles between 2 and 50 μm ; *Sand* (g
 774 kg^{-1}) is soil particles between 50 and 2000 μm ; P_{tot} is the total phosphorus in the soil layer ($g\ P\ kg^{-1}\ soil$); and N_{org} is
 775 the organic nitrogen of the soil layer ($g\ N\ kg^{-1}\ soil$).

776 There is a maximum soil P sorption capacity ($MaxPr_{tot}$, $mg\ P\ kg^{-1}\ soil$) in the model that is assumed to be related to
 777 the sum of oxalate-extractable Al and Fe ($(Al+Fe)_{ox}$, $mmol\ kg^{-1}\ soil$) (Schoumans & Groenendijk 2000):

$$778 \quad MaxPr_{tot} \approx 0.5(Al + Fe)_{ox} \cdot mw_P$$

779 (A4.8)

779 where mw_P is the molecular weight of P ($mg\ mmol^{-1}$).

780 The C_p in *ForSAFE* is solved using the following equation:

$$781 \quad \frac{\partial C_p}{\partial t} \cdot moist = Q_{in} \cdot Cp_{in} - C_p \left(Q + \frac{\partial moist}{\partial t} \right) + r_{des} + r_{wea} + r_{dep} + r_{min} + r_{upt}$$

782 (A4.9)

782 where Q_{in} and Q are the discharges of inflow and outflow of the soil layer, respectively ($dm^3\ water\ m^{-3}\ soil\ d^{-1}$);

783 Cp_{in} is the concentration of dissolved Pi in the inflow ($mg\ P\ dm^{-3}\ water$).