

1 **Impact of structural changes in wood-using industries on net carbon emissions in Finland**

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14

15 **ABSTRACT**

16

17 Forests and forest industries can contribute to climate change mitigation by sequestering carbon from

18 the atmosphere, by storing it in biomass and by fabricating products that substitute more greenhouse

19 gas emission intensive materials and energy. The objectives of the study are to specify alternative

20 scenarios for the diversification of wood products markets and to determine how an increasingly

21 diversified market structure could impact the net carbon emissions of forestry in Finland. The net

22 carbon emissions of the Finnish forest sector were modelled for the period 2016–2056 using a forest

23 management simulation and optimization model for the standing forests and soil, and separate models

24 for product carbon storage and substitution impacts. The annual harvest was fixed at approximately

25 70 Mm³, which was close to the level of roundwood removals for industry and energy in 2016. The

26 results show that the substitution benefits for a reference scenario with the 2016 market structure
27 account for 9.6 Mt C (35.2 Mt CO₂ equivalent [CO₂eq]) in 2056, which could be further increased by
28 7.1 Mt C (26 Mt CO₂eq) by altering the market structure. As a key outcome, increasing the use of by-
29 products for textiles and wood-plastic composites in place of kraft pulp and biofuel implies greater
30 overall substitution credits compared to increasing the level of log harvest for construction.

31

32

33 **Keywords:** bioeconomics; carbon emissions; forest products; industrial ecology; national forest
34 inventory; substitution

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37

38 INTRODUCTION

39

40 One of the core targets of the Paris Agreement is “*to achieve a balance between anthropogenic*
41 *emissions by sources and removals by sinks of greenhouse gases in the second half of this century*”
42 (UN 2015). The agreement also recognises the role of maintaining and managing forests towards this
43 objective. On one hand, forests contribute to climate change mitigation by sequestering carbon from
44 the atmosphere to trees and soil (Nabuurs et al. 2017). On the other hand, using harvested wood
45 products in place of more fossil emission intensive materials and energy sources can yield substitution
46 benefits in other economic sectors (Sathre and O’Connor 2010; Leskinen et al. 2018). It depends on
47 the scale of the substitution benefits and the carbon stocks of wood-based products, whether a trade-
48 off exists in the short-term between increasing wood harvesting and maintaining a larger net carbon
49 sink.

50

51 The global forest sector has experienced major changes in the 2000s (Hansen et al. 2013), and is
52 expected to do so on a pronounced scale in the future, partly driven by circular bioeconomy ambitions
53 (Hetemäki et al. 2017). Notably, graphic paper industries have been dwindling due to substitution by
54 electronic media, which has motivated forest industries to rethink their product portfolios (Näyhä and
55 Pesonen 2014; Hetemäki and Hurmekoski 2016). As a consequence, the European forest product
56 markets are no longer likely to follow the trends of the 20th century (Jonsson et al. 2017). To replace
57 some of the declining businesses, the wood using industries are expected to seek growth at least in
58 the construction, textile, chemical (including polymers), and liquid biofuel markets (Hurmekoski et
59 al. 2018). This could have important consequences for the overall GHG substitution benefit of wood
60 utilisation, as wood may increasingly replace other materials and energy carriers in the emerging
61 markets. Simultaneously, the use of wood is expected to decrease for graphic paper production, to
62 which no significant substitution benefit can be attributed (Achachlouei and Moberg 2015).

63

64 Assessing the net carbon emissions of wood use requires that the carbon fluxes in both the forest
65 ecosystem and related technosystem are quantified. The forest ecosystem emits CO₂ from respiration,
66 removes CO₂ from the atmosphere by photosynthesis, and stores carbon in above- and below-ground
67 biomass and soil. Harvested wood biomass is transferred to the technosystem, which stores carbon in
68 wood products, emits CO₂ from biomass combustion and fossil fuel based operations and avoids fossil
69 emissions due to material and energy substitution. In addition to CO₂ emissions, the forest ecosystem
70 and technosystem cause other greenhouse gas (GHG) emissions such as methane (CH₄) and nitrous
71 oxide (N₂O), and their contribution to climate change impacts can be meaningful. The climate impact
72 assessment of wood utilization can be simplified by converting all GHG emissions to CO₂ equivalents
73 on a 100-year time horizon. However, forests may have several other climate impacts than GHG
74 emissions, such as albedo, biogenic aerosols, effect on cloud formation, evaporation and surface
75 roughness (Kalliokoski et al. 2019).

76

77 In Finland, the carbon sequestration and stocks of boreal forests have both increased substantially
78 since the 1970's (Finnish Forest Research Institute 2014). This is especially because the mean annual
79 volume growth has clearly been greater than the mean annual amount of wood harvested (e.g. about
80 100 and 60 Mm³ yr⁻¹, respectively, in 2004-2013). Climate change and intensified forest management
81 have also contributed to this development (Hynynen et al. 2015; Henttonen et al. 2017; Finnish Forest
82 Research Institute 2014). Large uncertainties exist in the projected climate change (Ruosteenoja et al.
83 2016), which together with prevailing environmental conditions, forest structure, disturbance regime,
84 and forest management intensity, will likely affect the development of forests in different boreal
85 regions (Kellomäki et al. 2018; Heinonen et al. 2018a, 2018b).

86

87 The climate impacts of the forest-based bioeconomy have been assessed in several studies by
88 combining the carbon fluxes of the forest ecosystem and the technosystem. According to Heinonen
89 et al. (2017), the largest sustainable amount of domestic logging of sawlogs and pulpwood in Finland,
90 without endangering the stable wood supply, is 73 million m³ yr⁻¹ for the next 90-year period (not
91 considering the impacts of climate change). The total carbon balance of forestry was found to be the
92 highest with lower volumes of roundwood removal. On the other hand, with intensified forest
93 management (forest fertilisation, improved regeneration material and ditch network maintenance),
94 the sustainable harvest level could be increased by about 4 million m³ yr⁻¹ under a changing climate
95 over a period of 90 years (Heinonen et al. 2018a). Intensified management under moderate (RCP4.5)
96 climate change would make it possible to harvest about 80 million m³ yr⁻¹, without decreasing the
97 growing stock volume, over the next 90 years (Heinonen et al. 2018b).

98
99 Baul et al. (2017) calculated the net climate impacts for year 2016-2055 in southern Finland using
100 the currently recommended forest management scenario as a reference. In their study, maintaining
101 higher stocking with earlier final felling and the use of logging residues as energy appeared to be the
102 best option for increasing both the climate benefits and economic returns. On the other hand, the use
103 of alternative assumptions concerning GHG displacement potential largely affected the mitigation
104 potential of forest biomass. Soimakallio et al. (2016) found that an increased harvesting and use of
105 wood may not result in climate benefits for 100 years, due to the decline in the forest carbon sink.
106 Thus, there appears to be a significant trade-off between avoiding emissions through fossil fuel
107 substitution and reducing the forest carbon sink due to wood harvesting, despite a significant
108 compensation from substitution impacts. Seppälä et al. (2019) calculated a *required displacement*
109 *factor* for Finland, suggesting that in order to achieve net carbon emission reduction with increased
110 harvest, the average displacement factor of wood use should be between 2.0 and 2.4 tC/tC.

111

112 Previous studies have rarely analysed in depth how the changes in the market structure would
113 influence the substitution patterns and the net carbon emissions. The ambition of this study is to
114 highlight the impact of changing market structures and consequent substitution patterns, as this yields
115 a more realistic picture of the possible contribution of the substitution effect on the net carbon
116 emissions. Towards this backdrop, the objectives of the study are to i) define alternative market
117 structures for the future use of wood resources in Finland, based on the expected decline of graphic
118 paper markets and the growth of emerging wood-based products, ii) define displacement factors for
119 established and new wood-based products, weighted by the end uses of intermediate products; and
120 iii) based on these, determine how an increasingly diversified market structure may impact the net
121 carbon emissions of forestry in Finland.

122

123

124 **MATERIALS AND METHODS**

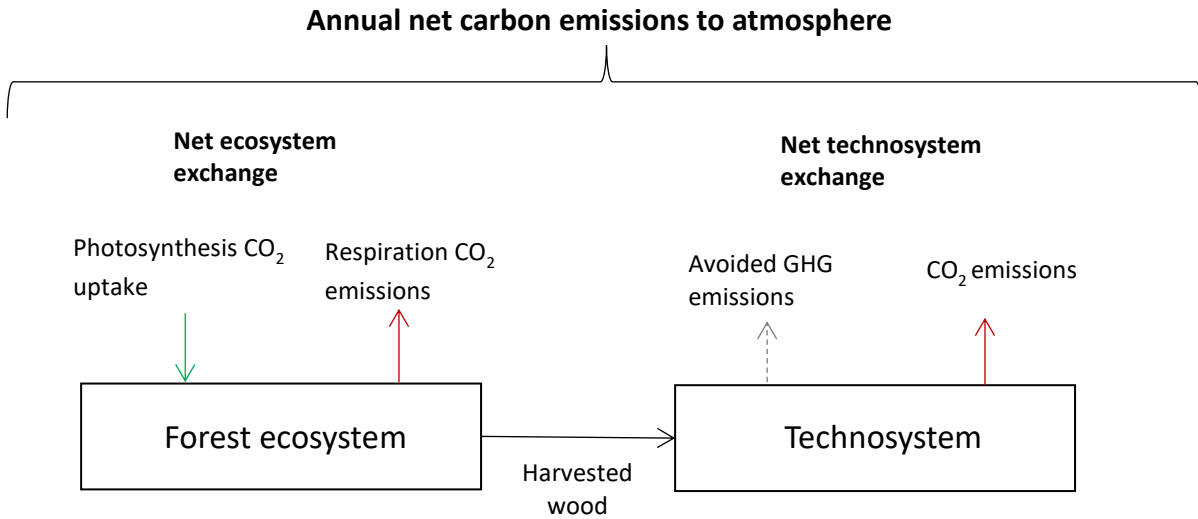
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126 *Computing the net carbon emissions for a forest-based production system*

127

128 The net carbon emissions (NCEs, in CO₂ equivalents) for a forest-based production system include
129 the carbon fluxes in the forest ecosystem and the related technosystem (Figure 1).

130



131

132 **Fig. 1.** Schematic figure of annual net carbon emissions to the atmosphere caused by carbon flows in
 133 a forest ecosystem and technosystem (Gustavsson et al. 2017).

134

135 The NCEs were calculated according to Equation 1:

136

137
$$NCE_t = (TC_{t-1} - TC_t) + (SC_{t-1} - SC_t) + (PC_{t-1} - PC_t) - SUB_P_t - SUB_EOL_t \quad (\text{Eq. 1})$$

138

139 , where TC = tree carbon stock, SC = soil carbon stock, PC = product carbon stock, SUB_P_t = avoided
 140 carbon emissions due to the substitution impact of harvested wood products (HWPs) at the production
 141 stage, SUB_EOL_t = avoided carbon emissions due to the substitution impact of HWPs at the end-of-
 142 life stage (utilisation as energy), and *t* = year.

143

144 The NCEs are reported as annual changes in the carbon stocks and as annual substitution impacts. A
 145 positive value of NCE translates to an increase in the net carbon emission to the atmosphere, while a
 146 negative value translates to the opposite.

147

148 ***Ecosystem carbon flows and assumptions***

149

150 ***Calculation of carbon flows in forests***

151

152 We calculated the carbon flows in forests (living forest biomass and soil) using Monsu software
153 (Pukkala 2011), which has been used in several scenario analyses of the impacts of forest
154 management and harvesting intensity on the timber supply and carbon balance of forestry (Heinonen
155 et al. 2018a, 2017, 2018b; Pukkala 2011, 2014, 2017; Zubizarreta-Gerendiain et al. 2016). The Monsu
156 software is able to calculate changes in all the carbon pools listed in the Intergovernmental Panel on
157 Climate Change's (IPCC's) carbon accounting rules, including: (1) living forest biomass; (2) soil
158 organic matter; and (3) wood-based products and fuels. In this study, however, the third component
159 was modelled separately.

160

161 The initial carbon pool for soil organic matter was calculated with models (Pukkala 2014) at the start
162 of the simulation. Tree species-specific turnover rates were used to calculate the litter production
163 from tree biomass (Pukkala 2014). The inputs to the soil carbon stock included dead organic matter
164 from litter, mortality, harvest residues (including tree tops, roots, branches, needles/leaves and bark)
165 and the growth of peat in undrained peatland forests. The release of carbon through the decomposition
166 of dead organic matter (CF_b) was simulated using the Yasso07 model (Tuomi et al. 2011b; Liski et
167 al. 2009; Tuomi et al. 2011a).

168

169 ***Forest data and the simulation of treatment schedules***

170

171 We used a sub-sample of the sample plots of the 11th National Forest Inventory (NFI11, 2009-2013)
172 of Finland (Korhonen 2016) as input data for the simulations. The forest data included one sample

173 plot from every inventory cluster. The plots were located on forestland assigned to timber production.
 174 A total of 1890, 1393 and 1402 plots were used for southern, central and northern Finland,
 175 respectively (Heinonen et al. 2018a, 2017, 2018b). Country-level results were aggregated from the
 176 separate analyses performed for these three regions.

177
 178 Different treatment schedules were simulated for each sample plot for ten 10-year periods, resulting
 179 in a 100-year simulation period (Table 1). A sample plot was managed with a particular treatment if
 180 the predefined conditions for such treatment were fulfilled in the middle of a 10-year period. In the
 181 simulations, the growth responses to climate change of different boreal tree species were considered
 182 by employing a new meta model approach (Seppälä et al. 2019). The model is based on the measured
 183 growth responses of Scots pine trees in provenance trials in Finland and Sweden (Beuker 1994;
 184 Persson and Beuker 1997; Berlin et al. 2016) and the growth responses predicted by an ecosystem
 185 model (Kellomäki et al. 2008, 2018). In the simulations, we assumed a mild climate change – the
 186 RCP2.6 forcing scenario of the Coupled Model Intercomparison Project 61 Phase 5 (CMIP5). In this
 187 scenario, the mean annual temperature is likely to increase by 2°C and annual mean precipitation by
 188 6% in Finland, if the atmospheric CO₂ concentration increases to 430 ppm by 2100 (Ruosteenoja et
 189 al. 2016).

190

191 **Table 1.** Simulated management activities at different sites.
 192

Forest regeneration	If the amount of natural advance regeneration was insufficient, sub-xeric sites were seeded with Scots pine, and mesic sites were planted either with Scots pine, Norway spruce or silver birch (probability 0.3, 0.6, 0.1), and herb-rich (or better) sites by Norway spruce or silver birch (probability 0.8 and 0.2). Other upland forest sites and all drained peatland sites were regenerated naturally. Seedlings were also expected to be born naturally at all sites, regardless of regeneration method. Sown seeds and planted seedlings had 10% higher growth rates than naturally born seedlings (Heinonen et al. 2018a, 2018b). Necessary regeneration treatments were performed immediately after clear cutting.
Tending treatments in seedling stands	Tending treatments for dense seedling and sapling stands were simulated according to Finnish management recommendations (Äijälä et al. 2014).

Thinning and final cutting	The earliest possible time for thinning (uniform) and final felling (clear cutting) were defined by multiplying currently recommended (Äijälä et al. 2014) lower threshold values for thinning (basal area) and final felling (diameter at breast height) by 0.9. Also treatment schedules with postponed cuttings and a schedule without cuttings were simulated (Heinonen et al. 2018a, 2018b).
Fertilisation	25% of those upland sub-xeric pine-dominated and mesic spruce-dominated stands were fertilised where the growing stock characteristics fulfilled a set of pre-determined criteria (Heinonen et al. 2018a, 2018b). Fertilisation was not simulated simultaneously (same 10-year period) with a thinning treatment.
Ditch network maintenance	On 40% of the drained peatlands, the ditch network was re-excavated to maintain the ditches when the growing stock characteristics fulfilled a set of pre-determined criteria (Heinonen et al. 2018a, 2018b).

193

194 ***Harvesting scenarios and optimisation***

195

196 In all wood-harvesting scenarios, the annual amount of harvested roundwood was fixed at 70 Mm³.
 197 This roughly corresponds to the realised annual harvest of sawlog and pulpwood in Finland in 2016
 198 (Natural Resources Institute of Finland 2017). In the reference scenario, the regional cutting targets
 199 for sawlog and pulpwood during the first five 10-year periods were derived for different tree species
 200 from the policy scenario in Lehtonen et al. (2016) separately for the three regions. The harvest targets
 201 of the last five periods were the same as during the fifth period.

202

203 The objective of the treatment scheduling problem was to maximise the timber production and
 204 profitability of forest management (net present value with a 3% discount rate), with species-specific
 205 even-flow harvesting targets for sawlog and pulpwood in each 10-year simulation period. More
 206 importance was given to fulfilling the harvesting targets of the earlier 10-year periods, if the cutting
 207 target could not be met during every period. The simulation and optimisation methods used in this
 208 study have been described in detail in Heinonen et al. (2018a, 2017, 2018b).

209

210 ***Technosystem carbon flows and assumptions***

211

212 **Scenarios**

213

214 Eight scenarios depicting plausible changes in the technosystem were defined (Table 2). All scenarios
215 were based on the same annual harvest level ($70 \text{ Mm}^3 \text{ yr}^{-1}$); that is, the scenarios only differed in
216 terms of market structure. The exact assumptions on market structures are described in supporting
217 information 1.

218

219 In Scenario 2 (Scenario 3), log production was increased (decreased) by 10% and pulpwood
220 production was decreased or increased by an equivalent wood amount to keep the same annual harvest
221 level. This was done because producing more sawlogs was expected to positively influence the net
222 carbon emissions (Pingoud et al. 2010), and because literature has emphasized the role of wood
223 construction in climate change mitigation (Smyth et al. 2017). These changes in sawlog and pulpwood
224 proportions were modelled by varying the top diameter threshold of sawlogs and by altering the
225 harvesting targets of sawlogs and pulpwood.

226

227 **Table 2.** Scenarios simulated in this study.

Scenario	Description
1. Reference	BaU (adopted from Lehtonen et al. (2016)) – Annual harvest in 2050 is 70 Mm^3
2. Sawlog +10	BaU, except for increased sawlog supply by 10% and decreased pulpwood supply by an equivalent share
3. Sawlog -10	BaU, except for decreased sawlog wood supply by 10% and increased pulpwood supply by an equivalent share
4. Biorefinery	Liquid biofuels and biochemicals account for 50% of all by-product use
5. Textiles	In addition to scenario 4, 50% of pulp production goes to dissolving pulp and further mostly to textiles
6. Composites	50% of pulp production is dissolving pulp (textiles); chemicals account for 50% of pulp side streams; composites and wood-based panels account for 50% of sawn wood residues
7. Graphic papers	BaU, except for assuming a negative DF (-0.58) for graphic paper production
8. Decarbonisation	BaU, except for assuming the average emissions of the energy sector to reduce by 80% by 2056 and consequently change the substitution impacts

228

229 Scenarios 4 to 6 were motivated by literature on emerging wood products, and the consequent
230 reallocation of wood flows in terms of by-product utilization (Hurmekoski et al. 2018; Kunttu et al.
231 2019). Currently, most sawmilling by-products are utilised in mill energy production, district heating
232 and electricity generation, while the rest is used for pulping. With the emerging renewable energy
233 technologies, increasing nuclear power, and improving energy efficiency, the pulping by-products
234 were assumed to be increasingly used for emerging wood-based products (Scenarios 4–6).
235 Additionally, in scenarios 4-6, communication paper production was assumed to decline towards
236 2050, due to digital media replacing print media. Similarly, the production of mechanical pulp was
237 assumed to have terminated by 2050, with Norway spruce pulpwood being used for kraft and
238 dissolving pulp instead.

239

240 Emerging biorefineries are expected to produce bioethanol and renewable diesel, and platform
241 chemicals to be refined to polymers and various other chemicals (Scenario 4). In terms of volume,
242 we assumed the chemicals to be drop-in or smart drop-in chemicals¹, for use as direct substitutes for
243 fossil-based ones. Dornburg et al. (2008) identified ethylene, used mostly for polyethylene (PE), as
244 being the most important bio-based intermediate chemical. It is also the largest of the currently
245 produced petrochemicals, by volume.

246

247 The production of dissolving pulp was assumed to grow significantly, driven by the increasing
248 demand for man-made cellulosic fibres for textiles (Scenarios 5 and 6) (Pöyry Inc. 2015). In addition,
249 in Scenario 6, a major shift from the use of by-products as energy to long-lived wood products (wood-
250 plastic composites and wood-based panels) was assumed (Kunttu et al. 2019).

¹ Bio-based drop-in chemicals such as ethylene are chemically identical to existing fossil-based chemicals. Smart drop-in chemicals refer to a special subgroup of drop-in chemicals such as succinic acid whose bio-based pathways provide advantages compared with the conventional petrochemical pathways, notably a comparably high biomass utilization efficiency, low embodied energy, noncomplex pathway, or low toxicity (Carus et al. 2017).

251

252 In addition to the market scenarios (1–6), two additional scenarios (7 and 8) were formulated to test
253 the sensitivity of the substitution impacts to some of the most important assumptions. Firstly, graphic
254 papers form one of the largest forest product categories by volume, yet only one single study could
255 be found that quantified the possible substitution impact between graphic paper and electronic media
256 (Achachlouei and Moberg 2015). To consider this uncertainty, we included one scenario assuming a
257 negative substitution impact for graphic papers (scenario 7).

258

259 Secondly, the emissions of the energy sector need to be reduced significantly towards 2050, to meet
260 the ambitious targets of the Paris Agreement. This is likely to reduce the relative climate benefit of
261 wood products (Peñaloza et al. 2018), as the emissions of wood-based products cannot be assumed
262 to decline at a similar pace as for the alternative products, because the fossil-based energy input for
263 the production of wood-based products in Finland is already relatively low. In an attempt to explore
264 the consequences of this major change, we estimated the impacts of decreasing the GHG emissions
265 from energy production for the displacement factors of production and end-of-life stages (Scenario
266 8).

267

268

269 ***Displacement factors***

270

271 The displacement factor, DF_i , for wood product i was calculated as:

272

$$273 \quad DF_i = \frac{GHG_{alternative} - GHG_{wood}}{WU_{wood} - WU_{alternative}} \quad (\text{Eq. 2})$$

274

275 , where $GHG_{alternative}$ and GHG_{wood} are the GHG emissions resulting from the use of the non-wood
276 and the wood alternatives expressed in mass units of carbon (C) corresponding to the CO₂ equivalent
277 of the emissions, and WU_{wood} and $WU_{alternative}$ are the amounts of wood used in the wood and non-
278 wood alternatives, expressed in mass units of C contained in the wood (Sathre and O'Connor 2010).
279

280 The GHG emissions in Eq. 2 were expressed as CO₂ equivalents in a 100-year time frame. The GHG
281 emissions were converted to a unit of carbon (C) by multiplying CO₂ equivalent (CO₂eq) by 12/44.
282 Also, the amounts of wood used were converted to carbon, resulting in a unit of tC/tC for the DF.
283 Here, we assumed a density of 460 kg/m³ for Scots pine, 410 kg/m³ for Norway spruce, and 580
284 kg/m³ for birch (Repola 2009). The carbon content of wood was assumed to be 50%. In cases where
285 DF was determined per total wood use, we used roundwood equivalent (RWE) conversion factors by
286 UNECE/FAO (2010) and Hurmekoski et al. (2018).

287
288 The DFs were determined mainly using publicly available domestic data sources, but in several cases
289 also other data sources and Ecoinvent database were utilized. Data for sawnwood and plywood DFs
290 were obtained from Vares and Häkkinen (2017), in which functionally similar wooden construction
291 elements' climate change impacts were compared to construction elements based on steel and
292 concrete. DFs were determined considering the mass of different construction elements used in a
293 representative building.

294
295 The DF for packaging was obtained from Knauf et al (2015). Packaging (container boards, carton
296 boards and sack paper) were assumed to replace plastic, glass and metal packaging (Knauf et al.
297 2015). Mechanical pulp was assumed to be used for magazine paper and no substitution credit was
298 given. However, the DF for graphic papers depends on a wide range of factors such as the brightness
299 of the screen of a mobile device and the number of readers, with one study suggesting that there could

300 be an average DF of -0.58, instead of the zero (Achachlouei and Moberg 2015). This value was used
301 for scenario 7.

302

303 DFs of diesel and ethanol were adapted from Bright et al. (2010) and Soimakallio et al. (2009).
304 Ethylene was assumed to ultimately replace plastics in packaging. Viscose, derived from dissolving
305 pulp, was assumed to replace cotton. The DF for textile replacement was adapted from the Ecoinvent
306 database 3.0 (see also supporting information 2). Wood-plastic composites (WPC) were assumed to
307 replace virgin polypropylene (PP) especially in car manufacturing (Judl et al. (2016), Ecoinvent
308 database 3.0).

309

310 Combined heat and power (CHP) was assumed to replace fossil energy. Wood biomass, such as chips,
311 residues and sawdust, are used typically mixed with other raw materials, especially peat. The
312 production of pulp consumes large amounts of energy, the majority of which is produced using the
313 by-products of pulp mills. As the generated energy was assumed to be used by the mills on site, no
314 substitution credit was assumed for mill energy.

315

316 For the end-of-life DFs, we assumed that all wood material products would be used for energy at the
317 end of their lifespan. This is currently a common practice in Finland and in most of the countries
318 where Finnish wood products and intermediate products are exported, as landfilling of organic and
319 biodegradable waste streams is forbidden. A widely used DF for energy displacement is 0.8 tC/tC
320 (Pukkala 2014). In this study we assumed that by 2016 energy displacement would be slightly smaller
321 due to the share of fossil energy having declined and the share of renewable energy increased in recent
322 years. Thus, the DF used for the EoL energy displacement was 0.7 tC/tC. For viscose, the end of life
323 energy DF was lower (0.23 tC/tC) as substituted cotton could be used as energy as well, but the energy

324 efficiency was considered lower compared to viscose. No EoL substitution credit was assigned for
325 chemicals.

326

327 In scenario 8, we assessed how the DFs and subsequent substitution impacts would change driven by
328 the decarbonization of the energy sector. Decarbonization influences both product displacement and
329 end of life energy displacement. We assumed that the GHG emissions of energy production would
330 be reduced to 20% of the level of emissions in 2016 by 2056, and re-estimated the DFs accordingly.
331 In line with this assumption, we assumed that only 20% of replaced energy would be fossil-based in
332 2056. Because of decarbonization, GHG emissions of processing of both wood products and their
333 substitutes would decrease in the future. Assumed decarbonization decreases the carbon footprints of
334 energy intensive products in particular. In order to determine new DFs because of decarbonization,
335 we determined the proportion of GHG emissions caused by energy use for all substituted and wood-
336 based products. Information on the share of energy production of the total climate change impacts of
337 plastics, steel and concrete were obtained from Material Economics Sverige AB (2018) and for other
338 products from Mattila et al. (2018). A linear decline in both the production stage and EoL DFs was
339 assumed, with the EoL DF declining from 0.7 tC/tC in 2016 to 0.14 tC/tC in 2056 (see Table 3).

340

341 In addition to the two extra scenarios (7 & 8) highlighting the uncertainties in DF estimates, we
342 carried out further sensitivity analysis for the DFs of textiles and WBCs by increasing and decreasing
343 their DF values by 50%. This was done, due to the reliability of the estimates remaining relatively
344 weak, and because they are important determinants of the overall net carbon emissions.

345

346 ***Substitution***

347

348 The wood use in Eq. 3 refers to the amount of wood embodied in the end product, as the wood
349 processing by-product flows were assigned to other markets, which were altered across scenarios.
350 The alternative product and wood product were functionally equivalent for each DF.

351

352 The substitution impact of the production stage for HWP (SUB_P, Equation 4) was calculated with
353 the help of DF_i and the production volume of wood product i (PV_i):

354

$$355 \quad SUB_P(t) = \sum DF_P_i(t) \cdot PV_i(t) \quad (\text{Eq. 4})$$

356

357 , where DF_P_i is the avoided fossil-based carbon emissions (tC) per carbon contained in product i
358 used (tC) in the production stage. This also includes substitution effects caused by the upstream
359 emissions of products.

360

361 In addition to material substitution, wood products can be used for energy substitution purposes in
362 their end-of-life stage. We assumed that wood products, for which the only possible end-of-life use
363 is combustion, would replace fossil fuels and merit a substitution credit (Table 3). The end-of-life
364 substitution impact was calculated according to Equation 5:

365

$$366 \quad SUB_EOL(t) = \sum DF_EOL_i(t) \times Outflow_i(t) \quad (\text{Eq. 5})$$

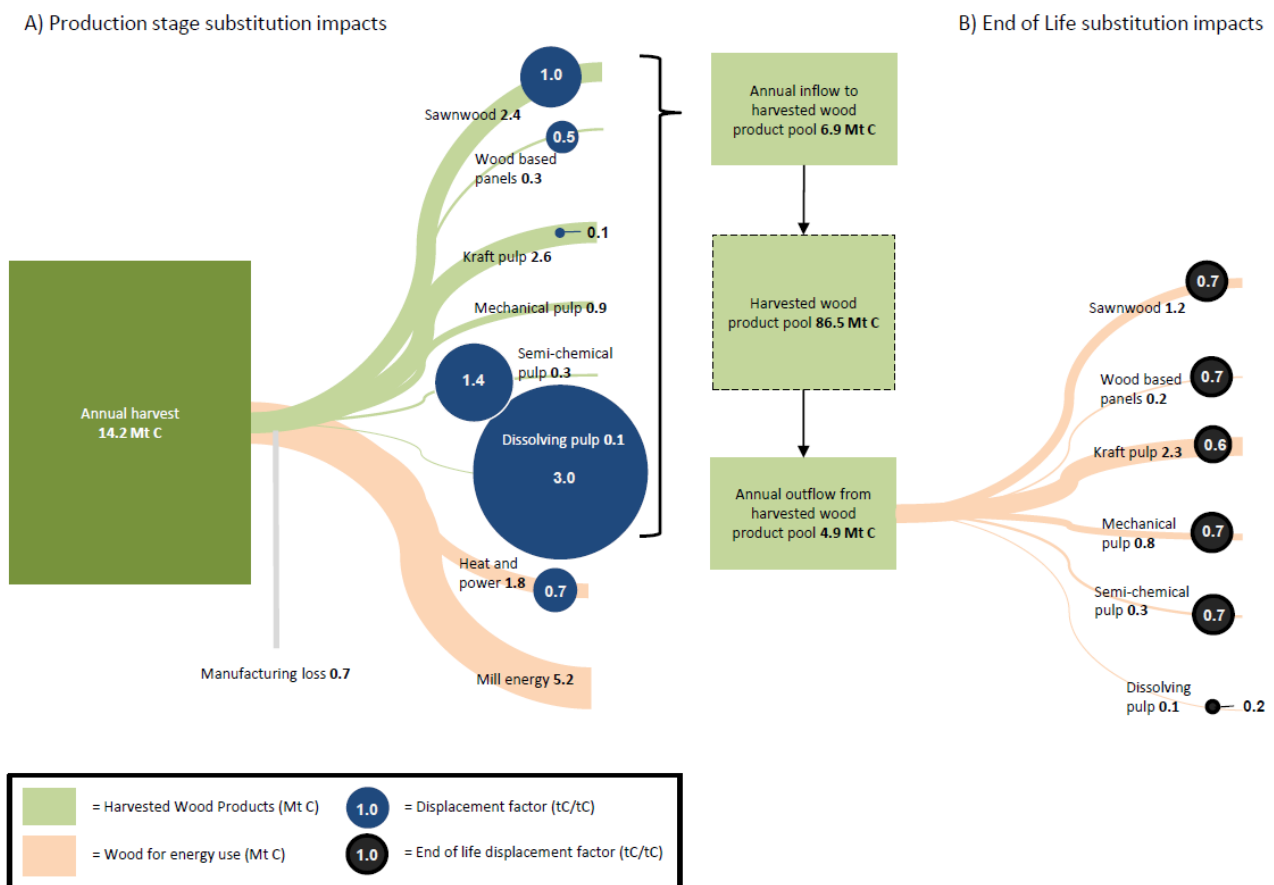
367

368 , where DF_EOL_i is the avoided fossil-based carbon emissions (tC) per carbon contained in product
369 i used (tC) for the end-of-life stage (incineration), and $Outflow_i$ is the outflow of wood product i from
370 the HWP pool (MtC yr⁻¹).

371

372 The total substitution impact of HWP is the sum of *SUB_P* and *SUB_EOL*. The average production
 373 stage displacement factors were calculated by summing up the production stage substitution impacts
 374 of all products and dividing the sum by the amount of carbon contained in total harvest (see also
 375 Figure 2, Table 3, and Supporting Information 3). The average end-of-life displacement factors were
 376 determined by dividing the end-of-life substitution impacts of all discarded wood products by the
 377 amount of carbon contained in the outflow from the HWP pool. We reported the average displacement
 378 factors for the production stage and the end-of-life stage separately, as the divisors are not necessarily
 379 comparable, in that one originates from annual harvest and the other from historical harvests as the
 380 HWP pool gradually decays.

381



382

383 **Figure 2.** Wood use allocation and respective weighted displacement factors estimated for 2016. Part
 384 (A) shows the amount of carbon (Mt C) in the wood used for each intermediate wood product and the
 385 respective weighted displacement factors. Part (B) shows the amount of carbon (Mt C) in discarded

386 wood products (outflow from the HWP pool) and the respective weighted end-of-life displacement
387 factors. The combined substitution impacts of (A) and (B), calculated as the product of wood flows
388 and weighted displacement factors, yields the total substitution impact in a given year.

389

390 The harvests were allocated for intermediate products that could be used for a range of end products.
391 To determine a weighted DF for each intermediate product, the end-use distribution of each
392 intermediate product was specified (Table 3). As no single data source was available, the end-use
393 shares were based on various sources (Poyry Inc 2017), and required assumptions particularly when
394 it came to the use of different pulp varieties for certain paper grades (Pöyry Inc. 2015). The level of
395 detail in the assumptions was limited by the complexity of the market and the consequent lack of
396 data. For example, due to a lack of more detailed or consistent data, we had to assume the same DFs
397 for the sawnwood produced from pine and spruce, despite the possibly differing end uses.

398

399 ***Product carbon storage***

400

401 The annual change in product carbon storage was calculated according to the IPCC's general model
402 called 'Product in Use' (Pingoud et al. 2006) in which estimates of the change in carbon are made by
403 tracking carbon inflows to, and outflows from, the 'products in use' carbon pool. The amounts of
404 inflow and outflow are influenced by product half-life periods (see Table 3). A product half-life is
405 the number of years it takes for one half of the material from the product stock to return into the
406 atmosphere. The carbon inflow to the pool was estimated by using the production volumes of different
407 HWP scenarios. The carbon pool was calculated from 1955 to 2056. The wood flows to 2016 were
408 estimated based on annual harvesting levels, and assuming the same production structure as in 2016.

409

410 We assumed a 5% manufacturing loss for all other products except for ethanol and ethylene. For these
 411 two products, the manufacturing loss was 88% and 93%, respectively, due to their very low biomass
 412 utilisation efficiency (Iffland et al. 2015). Some components could, in principle, be separated, and
 413 used to produce co-products, but due to a lack of robust data, and the possible variability between
 414 biorefineries, these possibilities were not considered. We assumed no direct roundwood flow to
 415 incineration; that is, only by-products could be used as bioenergy. Also logging residues were not
 416 within the system boundary of the used wood product model, i.e., no substitution credit or wood use
 417 were assigned to them.

418

419 Additional sensitivity analysis was carried out also for the calculation of the harvested wood product
 420 stock changes by reducing the HWP half-life coefficients by 50%. This would correspond, for
 421 example, to the estimate of Pingoud et al. (2001) stating that the average lifetime of buildings in
 422 Finland is less than 40 years. That is, earlier studies strongly suggest that the IPCC default coefficients
 423 are systematic overestimates, as there are substantial losses in wood material both in the construction
 424 phase and in the recovery of dismantled wood products for energy use, and because a higher share of
 425 wood products may end up in short-term uses or need to be replaced sooner than estimated (Statistics
 426 Finland 2010).

427

428 **Table 3.** End uses, displacement factors, and product half-life values of intermediate products.

429

Intermediate product	End use share	Displacement factor (DF_P) (tC/ tC) 2015	Displacement factor (DF_P) (tC/ tC) 2056 (scenario 8)	Half-life (years)
Solid wood products				
Sawnwood (incl. EWPs)	69 % construction	1.10	0.82	35
	19 % packaging (pallets)	1.10	-0.02	35
	3 % furniture	0.90	0.12	35
	9 % other	0	0	35
Plywood	41 % construction	1.10	0.74	35

	59 % other	0	0	35
Pulp and paper				
Mechanical pulp	100 % graphic papers (incl. newsprint)	0	0	2
Semi-chemical pulp	100 % packaging (container boards)	1.40	0.30	2
Kraft pulp	38 % graphic papers	0	0	2
	45 % hygienic: tissue, toilet, etc. (incl. fluff pulp)	0	0	2
	8 % packaging (carton boards, sack paper)	1.40	0.30	2
Dissolving pulp	74 % Textiles (viscose)	4.00	2.00	2
	26 % other	0	0	2
Heat and power				
CHP	100 % energy	0.70 ¹	0.14	1
mill energy	100 % energy	0 ²	0	1
Biorefining				
diesel	100 % transport fuel	0.63	N.A. ³	2
ethanol	100 % transport fuel	0.70	N.A.	2
ethylene	100 % packaging (PE, PET)	1.3	N.A.	2
Composites				
wood-plastic composite	70 % terraces	0	0	
	30 % plastic components for cars (polypropylene)	7.38	3.20	35

430

¹ assuming fossil energy substitution

431

² no substitution credits for mill energy has been given as the energy is used for fuelling the production process of wood-based products.

432

433

³ N.A. = Not applicable (data not used)

434

435

436

RESULTS

437

438

The scenarios altering the ratio of sawlog and pulpwood supply had a minor impact on carbon stock

439

changes in forests, soil and products (Fig. 3). The only exception was the standing tree biomass in the

440

period of 2046-2056. This can be explained by a temporary deceleration of tree growth when targeting

441

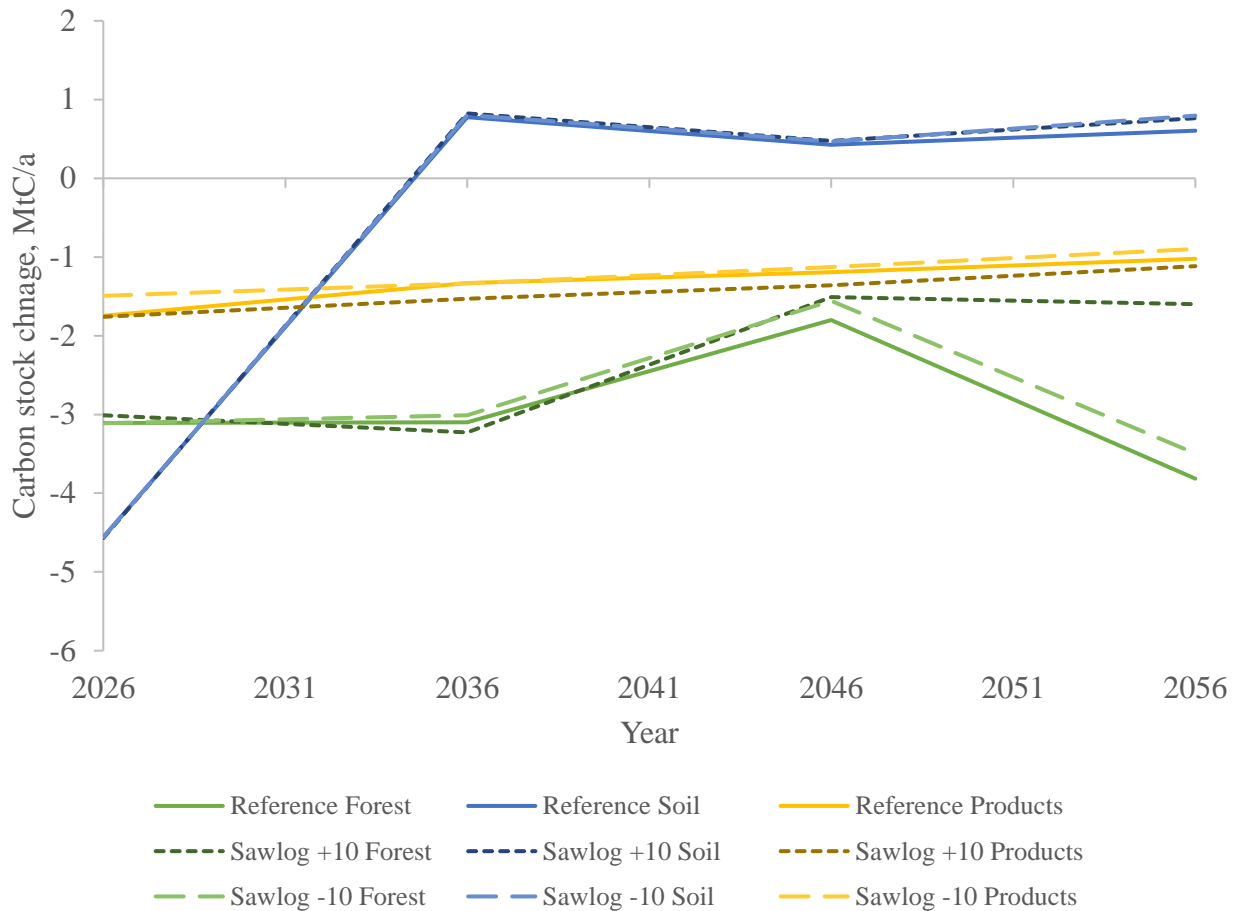
an increased harvest of sawlogs. The peak in soil carbon at the beginning of the simulation period

442

was likely due to an immediate intensification of young forest management, which caused a surge of

443

rapidly decaying litter to the soil.



445

446 **Figure 3.** Carbon stock change in forest, soil and products in scenarios 1–3. Underlying data used
 447 to create this figure can be found in the supporting information 4.
 448

449 The scenarios showed vast differences in the substitution impact (Table 4). The average production
 450 stage displacement factors in 2056 ranged from a minimum of 0.3 tC/tC (Scenario 3 – log decrease
 451 and scenario 4 - biorefinery) to a maximum of 0.8 tC/tC (Scenario 6 - composites). The scenarios
 452 showed that altering the ratio of sawlog and pulpwood production had a relatively minor impact
 453 (Scenarios 2-3) compared to altering the end-use of pulp from graphic papers to textiles (Scenario 5)
 454 and the use of by-products from energy to long-lived products (scenario 6). This was partly due to a
 455 relatively minor alteration in the sawlog and pulpwood ratio which significantly limited the increase
 456 in the production of solid wood products. The biorefinery scenario (4) showed slightly lower
 457 substitution benefits compared to the baseline, which was mainly due to the extremely low efficiency

458 of feedstock conversion to ethanol and ethylene, and the assumed lack of end-of-life substitution
 459 gains when compared to many paper products.

460

461 **Table 4.** Substitution impact (production and end-of-life) and average displacement factors at the end
 462 of the simulation period (in 2056).

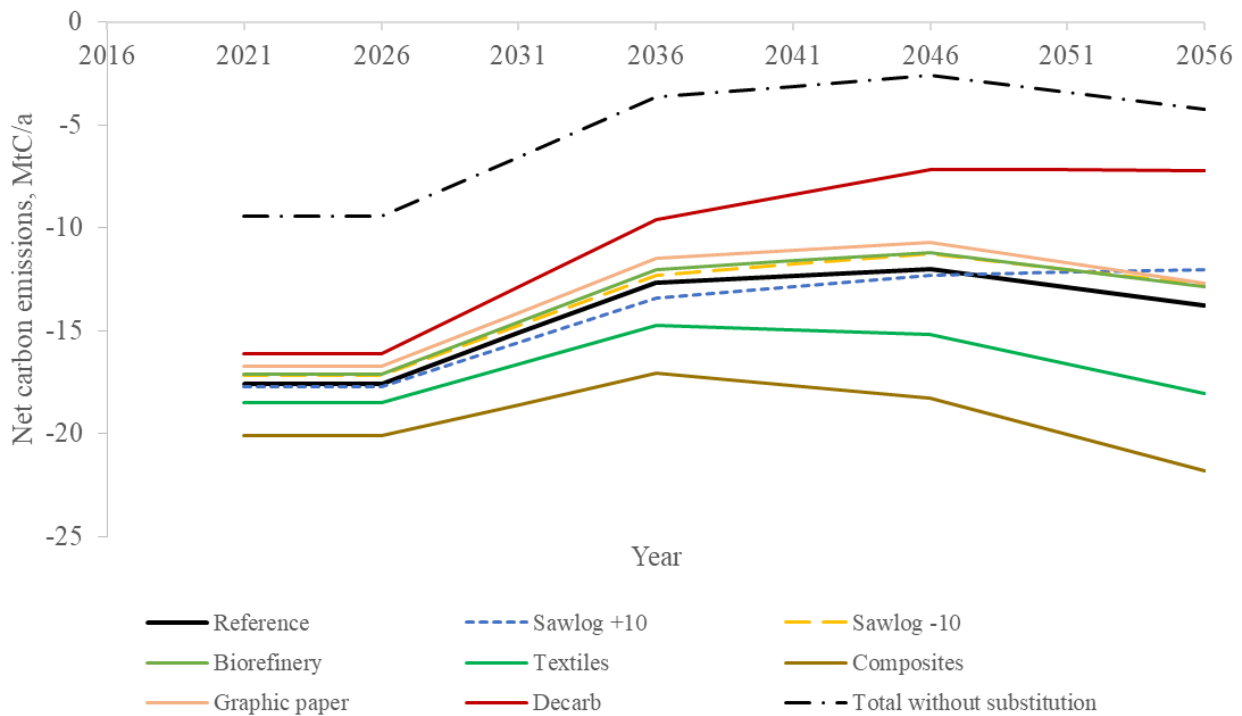
463

	Total substitution, MtC	Average DF - production	Average DF - EoL
1. Reference	-9.6	0.33	0.63
2. Sawlog +10	-10.1	0.35	0.64
3. Sawlog -10	-9.2	0.30	0.63
4. Biorefinery	-8.7	0.30	0.36
5. Textiles	-13.0	0.60	0.30
6. Composites	-16.8	0.80	0.35
7. Graphic paper	-8.4	0.25	0.63
8. Decarbonisation	-3.0	0.14	0.13

464

465 With all of the carbon flows considered, the Finnish forestry sector remained a net carbon sink in all
 466 scenarios (Fig. 4). The harvest rate of 70 Mm³ yr⁻¹ first reduces the net sink, but in some scenarios
 467 this is compensated for by an increase in forest growth, or in substitution impacts. The difference in
 468 the net carbon emissions between the reference scenario and Scenario 6 (composites) in 2056 equalled
 469 8.1 MtC. In comparison, the difference between the net carbon emission with and without considering
 470 the substitution benefits for the reference scenario was around 9.5 MtC. The substitution impact
 471 (including the production and end-of-life stages) constituted 42–84% of the total net carbon
 472 emissions. This was partly a result of a stable harvest level over the projection period that was close
 473 to the maximum sustainable harvest.

474



475
 476 **Figure 4.** Net carbon emissions (carbon stock changes in soil, forest and products, and avoided carbon
 477 emissions due to substitution). For comparison, the reference scenario without substitution impact is
 478 shown. Underlying data used to create this figure can be found in the supporting information 4.
 479

480 Compared to the reference scenario, only Scenarios 5 (textiles) and 6 (composites) resulted in a
 481 reduction in the net carbon emissions by 2056. The rest of the scenarios suggested a relatively minor
 482 impact of changing market or logging structure on the net carbon emissions of Finnish forestry at the
 483 end of the projection period. Even assuming the graphic papers to have a negative substitution impact
 484 (Scenario 7) resulted in a very similar overall substitution impact compared to the biorefinery scenario
 485 (Scenario 4). Only the scenario (8) assuming greatly diminishing displacement factors due to rapidly
 486 reducing emissions in the energy sector would lead to a significant reduction in substitution benefits
 487 and a consequent increase in net carbon emissions.

488
 489 Further sensitivity analysis was carried out for both the displacement factors and the HWP half-life
 490 coefficients. The substitution impacts of the textiles scenario (5) changed $\pm 30\%$ in 2056, when the
 491 textile DF was increased/decreased by 50%, whereas the impacts of composites scenario (6) changed
 492 $\pm 10\%$ when the composite DF was changed by 50%. The impact of DF estimates is clearly large, yet

493 even a significant change in them would not alter the main conclusions derived from the results. That
494 is, the change of $\pm 50\%$ in the DFs did not change the ranking of the scenarios in terms of the net
495 carbon emissions. When reducing the HWP half-life coefficients by 50%, the overall substitution
496 impacts increased by 2.9-4.3% (3.9% for the reference scenario), while the net carbon emissions
497 declined by 1.1-2.5% (1.3% for the reference scenario). That is, while end-of-life substitution benefits
498 were increased due to an increase in the annual HWP pool outflow, the shortened life span of wood
499 products had a greater opposite impact on net emissions. The impact of changes in half-life
500 coefficients on results was overall small, and did not change the ranking of the scenarios.

501

502

503 **DISCUSSION**

504

505 This study examined the impact of a changing market structure in the wood-using industries on the
506 net carbon emissions in Finland. The scenarios presented in the study are not forecasts, but they could
507 be taken as alternative options to facilitate ‘what if’ analysis on the possible impact of changing
508 market structures on net emissions and displacement factors for wood-based products.

509

510 According to the results, it is possible that the substitution benefits of wood use could be significantly
511 increased or decreased, as compared to the current market structure. The difference in the net carbon
512 emissions between the reference scenario and the scenario with the highest substitution impact
513 (increased production of wood-plastic composites and textiles) in 2056 was 8.1 Mt C (29.7 Mt
514 CO₂eq). In comparison, the difference between the net carbon emission with and without considering
515 the substitution benefits for the reference scenario was around 9.5 Mt C (34.8 Mt CO₂eq). These
516 values compare to total emissions (excluding the LULUCF sector) of 58.1 Mt CO₂eq in Finland in
517 2016.

518

519 When considering all industrial uses of wood, the average production stage displacement factor in
520 the reference scenario was 0.33 tC/tC at the end of the simulation period (0.34 in 2016), and ranged
521 from a minimum of 0.3 to a maximum of 0.8. Earlier studies have reported weighted displacement
522 factors of around 0.5 tC/tC for the production stage for sawnwood, panels, and energy (Suter et al.
523 2017; Smyth et al. 2017).

524

525 By altering the wood use assumptions, it appears that increasing the substitution benefits is more
526 likely to be accomplished by shifting the use of by-products from kraft pulp and heat and power
527 production to textile, composite and wood-based panel production than by increasing sawn wood
528 production. In addition, the results suggest that the end uses of pulp and by-products were more
529 decisive compared to the ratio of sawn wood and pulp production. Although the scope of assumptions
530 of this study may influence the conclusions, it seems the end-use distribution of intermediate products
531 were more decisive for the substitution impact than the volume of intermediate-product production,
532 given a fixed overall harvest level. Further, while the sensitivity analysis on the displacement factors
533 showed significant variation in the results, the changes in the allocation of harvest to different end
534 uses, i.e., the scenarios themselves, appeared to have the most influence. This said, apart from the
535 relatively established DFs for construction, the DFs in particular for the emerging end uses of wood
536 remain highly uncertain, and possibly not fully controlled by the sensitivity analysis carried out in
537 this study. Moreover, there might be remarkable losses in the recovery of discarded wood products
538 for energy use at the end-of-life and there might be similar inefficiencies in the system not captured
539 by the assumed 5% manufacturing loss. Consequently, the overall substitution benefit may be lower
540 than estimated.

541

542 Due to significant knowledge gaps regarding the emerging uses of wood, there is a need for balancing
543 between expected developments in the forest products markets and the availability of data for the
544 determination of displacement factors. For some of the emerging wood-based products, such as
545 alternative solvent processes for regenerated cellulose fibres for textiles, the displacement factors
546 could turn out to be higher than assumed in this study. Similarly, using lignin as a concrete admixture
547 reduces the need for cement and water in concrete production. This may, in principle, be an effective
548 strategy for emission reductions in the construction sector, in addition to substituting concrete for
549 sawn wood. While such possibilities remain speculative, they do demonstrate that the range of
550 substitution impacts depicted in this study does not necessarily represent the maximum plausible
551 range. Our analysis also disregards the possible uptake of carbon-capture technology (e.g., BECCS)
552 which features visibly, for example, in the IPCC 1.5 degree special report (IPCC 2018), and which,
553 if integrated into a biorefinery, could profoundly alter our conclusions.

554

555 DFs can generally be expected to decrease over time, as all sectors of the economy are required to
556 meet climate change mitigation obligations, which is likely to reduce fossil energy use and increase
557 renewable energy sources and energy efficiency (Keith et al. 2015). Another highly relevant yet not
558 well understood factor is the influence of increasing recycling rates of non-wood products on the
559 wood-product displacement factors. In a case where a circular economy decreases the emissions of
560 substitute products, the relative substitution benefit of wood-based products is likely to be reduced,
561 such as in the case of replacing recycled plastic compared to primary plastic. The current study
562 quantified only the approximate overall impact of decreasing the emissions of the energy sector. Our
563 results show that the substitution benefit may greatly diminish, yet not completely vanish, assuming
564 reduced emissions from the energy sector. Nonetheless, as noted by Peñaloza et al. (2018), the priority
565 ought to be to substitute emission-intensive materials using several different approaches
566 simultaneously to maximise the overall climate change mitigation efforts. Further research in the

567 context of substitution and displacement factors could therefore aim to prioritise such end uses where
568 significant emissions reductions in the existing regime are the most difficult to gain.

569

570 Besides the displacement factors, there are considerable uncertainties related to HWP half-lives,
571 although their impact on the net carbon emissions is more marginal. In the absence of more consistent
572 data, the half-lives applied in this study were based on widely cited IPCC reports (Pingoud et al. 2006;
573 Skog 2003). However, as pointed out by several authors, also these values can be criticised (e.g.
574 (Jordan et al. 2018)). While the impact of the half-life assumptions in the scenarios studied remained
575 minor, their relative impact ought to be growing due to the general downward trend of the substitution
576 impacts.

577

578 Irrespective of the scenario, the Finnish forestry sector was projected to remain a net carbon sink with
579 a constant harvest level of 70 Mm³ yr⁻¹. Further conclusion on the net carbon emissions is sensitive
580 to initial values and the applied reference (counterfactual) scenario. In this study, different market
581 scenarios were only contrasted against a fixed market structure, as opposed to varying harvest
582 intensities, as the simulated harvest level already mirrored the level of industrial wood harvest in
583 2016. Comparing the net carbon emissions against a selected reference scenario, in which, for
584 example, no industrial utilisation of forests takes place or in which less intensive harvests are made
585 in the next decades, would significantly affect the conclusions drawn. In one such study (Soimakallio
586 et al. 2016), a substitution credit of 12.4 was not sufficient to compensate for the reduction in forest
587 carbon sink (14.6 MtC) on a 100-year time horizon. Furthermore, Seppälä et al. (2019) estimated that
588 domestic wood harvest of 77 Mm³ yr⁻¹ instead of 57 Mm³ yr⁻¹ in Finland would mean that the average
589 DF for increased wood use should be 2–2.4 tC/tC in timeframe 2017-2116 in order to achieve net
590 reductions in GHG emissions. This clearly exceeds the 0.3-0.8 attained in the current study. On the

591 other hand, the expected increase in various damage risks to forests could at least partially cancel the
592 predicted increase in forest productivity under a changing climate (Reyer et al. 2017; Kauppi et al.
593 2018), which the study did not account for. That is, although the global warming has so far increased
594 the growth of Finnish forests, natural disturbances such as storms, insect outbreaks and forest fires
595 are estimated to substantially increase, which could again change the outcome. Further research is
596 needed to illuminate the nature and extent of the apparent trade-off between forest carbon sinks and
597 wood product substitution impacts, using, for example, the carbon balance indicator developed by
598 Pingoud et al. (2016), and by considering also uncertainties related to climate change and its impacts
599 to forests and forestry.

600

601 Finally, it is important to stress that climate policy recommendations regarding the forest sector
602 should not be drawn based only on the results concerning the forest sink, industry structure and the
603 displacement factor. More holistic analysis would be needed to additionally consider, for example,
604 the long-term forest growth dynamics, the adaptation of forests to changing climate and the increasing
605 forest disturbances, the albedo and aerosol impacts of forests, and the international carbon leakage
606 effects (Seidl et al. 2017; Kauppi et al. 2018; Kalliokoski et al. 2019; Kallio and Solberg 2018). Thus,
607 significant challenges remain for quantifying the overall climate impact of the industrial use of forests
608 in a policy relevant manner, i.e., without restricting the system boundaries.

609

610

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612

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620

621 **REFERENCES**

- 622 Achachlouei, M.A. and Å. Moberg. 2015. Life cycle assessment of a magazine, part II: A
623 comparison of print and tablet editions. *Journal of Industrial Ecology* 19(4): 590–606.
- 624 Äijälä, O., A. Koistinen, J. Sved, K. Vanhatalo, and P. Väisänen. 2014. Hyvän metsänhoidon
625 suositukset—Metsänhoito (Recommendations for good forest management—forest
626 management). *Forestry Development Centre Tapio, Helsinki, Finland.*[In Finnish.].
- 627 Baul, T.K., A. Alam, A. Ikonen, H. Strandman, A. Asikainen, H. Peltola, and A. Kilpeläinen. 2017.
628 Climate change mitigation potential in boreal forests: Impacts of management, harvest
629 intensity and use of forest biomass to substitute fossil resources. *Forests* 8(11): 455.
- 630 Berlin, M., T. Persson, G. Jansson, M. Haapanen, S. Ruotsalainen, L. Barring, and B. Andersson
631 Gull. 2016. Scots pine transfer effect models for growth and survival in Sweden and Finland.
- 632 Beuker, E. 1994. Long-term effects of temperature on the wood production of *Pinus sylvestris* L.
633 and *Picea abies* (L.) Karst. in old provenance experiments. *Scandinavian Journal of Forest*
634 *Research* 9(1–4): 34–45.
- 635 Bright, R.M., A.H. Strømman, and T.R. Hawkins. 2010. Environmental Assessment of Wood-
636 Based Biofuel Production and Consumption Scenarios in Norway. *Journal of Industrial*
637 *Ecology* 14(3): 422–439.
- 638 Carus, M., L. Dammer, Á. Puente, A. Raschka, and O. Arendt. 2017. *Bio-based drop-in, smart*
639 *drop-in and dedicated chemicals*. NOVA Institute.
- 640 Dornburg, V., B.G. Hermann, and M.K. Patel. 2008. Scenario projections for future market
641 potentials of biobased bulk chemicals. *Environ. Sci. Technol.* 42: 2261–2267.
- 642 Finnish Forest Research Institute. 2014. *Statistical Yearbook of Forestry 2014*. Helsinki, Finland.
- 643 Gustavsson, L., S. Haus, M. Lundblad, A. Lundström, C.A. Ortiz, R. Sathre, N. Le Truong, and P.-
644 E. Wikberg. 2017. Climate change effects of forestry and substitution of carbon-intensive
645 materials and fossil fuels. *Renewable and Sustainable Energy Reviews* 67: 612–624.

- 646 Hansen, E., R. Panwar, and R. Vlosky. 2013. *The Global Forest Sector: Changes, Practices, and*
647 *Prospects*. CRC Press.
- 648 Heinonen, T., T. Pukkala, A. Asikainen, and H. Peltola. 2018a. Scenario analyses on the effects of
649 fertilization, improved regeneration material, and ditch network maintenance on timber
650 production of Finnish forests. *European Journal of Forest Research* 137(1): 93–107.
- 651 Heinonen, T., T. Pukkala, S. Kellomäki, H. Strandman, A. Asikainen, A. Venäläinen, and H.
652 Peltola. 2018b. Effects of forest management and harvesting intensity on the timber supply
653 from Finnish forests in a changing climate. *Canadian Journal of Forest Research* 48(999): 1–
654 11.
- 655 Heinonen, T., T. Pukkala, L. Mehtätalo, A. Asikainen, J. Kangas, and H. Peltola. 2017. Scenario
656 analyses for the effects of harvesting intensity on development of forest resources, timber
657 supply, carbon balance and biodiversity of Finnish forestry. *Forest Policy and Economics* 80:
658 80–98.
- 659 Henttonen, H.M., P. Nöjd, and H. Mäkinen. 2017. Environment-induced growth changes in the
660 Finnish forests during 1971–2010—An analysis based on National Forest Inventory. *Forest*
661 *Ecology and Management* 386: 22–36.
- 662 Hetemäki, L., M. Hanewinkel, B. Muys, M. Ollikainen, M. Palahí, and A. Trasobares. 2017.
663 *Leading the way to a European circular bioeconomy strategy*. From Science to Policy 5.
664 European Forest Institute.
- 665 Hetemäki, L. and E. Hurmekoski. 2016. Forest Products Markets under Change: Review and
666 Research Implications. *Current Forestry Reports* 2: 177–188.
- 667 Hurmekoski, E., R. Jonsson, J. Korhonen, J. Jänis, M. Mäkinen, P. Leskinen, and L. Hetemäki.
668 2018. Diversification of the forest-based sector: Role of new products. *Canadian Journal of*
669 *Forest Research* 48(12): 1417–1432.
- 670 Hynynen, J., H. Salminen, A. Ahtikoski, S. Huuskonen, R. Ojansuu, J. Siipilehto, M. Lehtonen, and

671 K. Eerikäinen. 2015. Long-term impacts of forest management on biomass supply and forest
672 resource development: a scenario analysis for Finland. *European Journal of Forest Research*
673 134(3): 415–431.

674 Iffland, K., M. Carus, F. de Bie, L. Diels, J. van Haveren, P. Willems, J. Ravenstijn, E. Vink, and K.
675 Wagemann. 2015. Definition, Calculation and Comparison of the “Biomass Utilization
676 Efficiencies (BUE)” of Various Bio-based Chemicals, Polymers and Fuels.

677 Iordan, C.-M., X. Hu, A. Arvesen, P. Kauppi, and F. Cherubini. 2018. Contribution of forest wood
678 products to negative emissions: historical comparative analysis from 1960 to 2015 in Norway,
679 Sweden and Finland. *Carbon Balance and Management* 13(1): 12.

680 IPCC. 2018. *Global warming of 1.5°C. An IPCC Special Report on the impacts of global warming*
681 *of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in*
682 *the context of strengthening the global response to the threat of climate change,.*
683 Intergovernmental Panel on Climate Change.

684 Jonsson, R., E. Hurmekoski, L. Hetemäki, and J. Prestemon. 2017. What is the current state of
685 forest product markets and how will they develop in the future? In *Towards a Sustainable*
686 *European Forest-Based Bioeconomy – Assessment and the Way Forward*, ed. by G Winkel,
687 126–131. Grano Oy, Joensuu, Finland: What Science Can Tell Us, European Forest Institute.

688 Judl, J., M. Hildén, R. Antikainen, A. Temmes, M. Kuisma, and P. Peck. 2016. *The renewal of*
689 *forest-based industries needs to focus on environmental opportunities and challenges.*
690 *Renewal of manufacturing* 10/2016.

691 Kallio, A.M.I. and B. Solberg. 2018. Leakage of forest harvest changes in a small open economy:
692 case Norway. *Scandinavian Journal of Forest Research* 33(5): 502–510.

693 Kalliokoski, T., T. Aalto, J. Bäck, E. Ezhova, D. Franz, S. Haapanala, E. Juurola, et al. 2019.
694 *Carbon sink and CarbonSink+: from observations to global potential.* University of Helsinki.
695 <https://researchportal.helsinki.fi/en/publications/carbon-sink-and-carbonsink-from->

696 observations-to-global-potential.

697 Kauppi, P., M. Hanewinkel, L. Lundmark, G. Nabuurs, H. Peltola, A. Trasobares, and L. Hetemäki.
698 2018. *Climate Smart Forestry in Europe*. European Forest Institute.

699 Keith, H., D. Lindenmayer, A. Macintosh, and B. Mackey. 2015. Under what circumstances do
700 wood products from native forests benefit climate change mitigation? *PloS One* 10(10):
701 e0139640.

702 Kellomäki, S., H. Peltola, T. Nuutinen, K.T. Korhonen, and H. Strandman. 2008. Sensitivity of
703 managed boreal forests in Finland to climate change, with implications for adaptive
704 management. *Philosophical Transactions of the Royal Society of London B: Biological
705 Sciences* 363(1501): 2339–2349.

706 Kellomäki, S., H. Strandman, T. Heinonen, A. Asikainen, A. Venäläinen, and H. Peltola. 2018.
707 Temporal and Spatial Change in Diameter Growth of Boreal Scots Pine, Norway Spruce, and
708 Birch under Recent-Generation (CMIP5) Global Climate Model Projections for the 21st
709 Century. *Forests* 9(3): 118.

710 Knauf, M., M. Köhl, V. Mues, K. Olschofsky, and A. Frühwald. 2015. Modeling the CO₂-effects
711 of forest management and wood usage on a regional basis. *Carbon Balance and Management*
712 10(1): 13.

713 Korhonen, K.T. 2016. National forest inventories: Assessment of wood availability and use:
714 Finland. *National Forest Inventories: Assessment of Wood Availability and Use/Eds. C. Vidal,
715 I. Alberdi, L. Hernandez and JJ Redmond*.

716 Kunttu, J., E. Hurmekoski, H. Heräjärvi, T. Hujala, and P. Leskinen. 2019. Preferable utilisation
717 patterns of wood product industries' by-products in Finland. *Forest Policy and Economics* In
718 press.

719 Lehtonen, A., O. Salminen, M. Kallio, T. Tuomainen, and R. Sievänen. 2016. Skenaariolaskelmiin
720 perustuva puuston ja metsien kasvihuonekaasutaseen kehitys vuoteen 2045: Selvitys maa-ja

721 metsätalousministeriölle vuoden 2016 energia-ja ilmastostrategian valmistelua varten.

722 Leskinen, P., G. Cardellini, S. Gonzalez-Garcia, L. Gustavsson, E. Hurmekoski, R. Sathre, J.
723 Seppälä, C. Smyth, T. Stern, and H. Verkerk. 2018. *Substitution effects of wood-based*
724 *products in climate change mitigation*. From Science to Policy, European Forest Institute.
725 Manuscript.

726 Liski, J., M. Tuomi, and J. Rasinmäki. 2009. Yasso07 user-interface manual. *Finnish Environment*
727 *Institute, Helsinki* 615.

728 Material Economics Sverige AB. 2018. *The circular economy - a powerful force for climate*
729 *mitigation. Transformative innovation for prosperous and low-carbon industry*. Stockholm,
730 Sweden.

731 Mattila, T.J., N. Järviö, and J. Seppälä. 2018. Climate mitigation in the energy sector reduces
732 potential climate potential of increased wood use. *Submitted to Journal of Industrial Ecology*.

733 Nabuurs, G.-J., P. Delacote, D. Ellison, M. Hanewinkel, L. Hetemäki, M. Lindner, and M.
734 Ollikainen. 2017. By 2050 the mitigation effects of EU forests could nearly double through
735 climate smart forestry. *Forests* 8(12): 484.

736 Natural Resources Institute of Finland. 2017. Wood in energy generation in 2016.
737 <http://stat.luke.fi/en/wood-energy-generation>.

738 Näyhä, A. and H.-L. Pesonen. 2014. Strategic change in the forest industry towards the biorefining
739 business. *Technological Forecasting and Social Change* 81: 259–271.

740 Peñaloza, D., M. Erlandsson, J. Berlin, M. Wålinder, and A. Falk. 2018. Future scenarios for
741 climate mitigation of new construction in Sweden: Effects of different technological pathways.
742 *Journal of Cleaner Production* 187: 1025–1035.

743 Persson, B. and E. Beuker. 1997. Distinguishing between the effects of changes in temperature and
744 light climate using provenance trials with *Pinus sylvestris* in Sweden. *Canadian Journal of*
745 *Forest Research* 27(4): 572–579.

- 746 Pingoud, K., T. Ekholm, S. Soimakallio, and T. Helin. 2016. Carbon balance indicator for forest
747 bioenergy scenarios. *Gcb Bioenergy* 8(1): 171–182.
- 748 Pingoud, K., A.-L. Perälä, and A. Pussinen. 2001. Carbon dynamics in wood products. *Mitigation
749 and Adaptation Strategies for Global Change* 6(2): 91–111.
- 750 Pingoud, K., J. Pohjola, and L. Valsta. 2010. Assessing the integrated climatic impacts of forestry
751 and wood products. *Silva Fennica* 44(1): 155–175.
- 752 Pingoud, K., K. Skog, and D. Have. 2006. Harvested wood products.
- 753 Pöyry Inc. 2015. *World fibre outlook up to 2030*. Finland: Vantaa.
- 754 Poyry Inc. 2017. *Metsäbiomassan kustannustehokas käyttö [Cost-effective use of forest biomass]*.
755 Valtioneuvoston selvitys- ja tutkimustoiminnan julkaisusarja 23/2017.
- 756 Pukkala, T. 2011. Optimizing forest management in Finland with carbon subsidies and taxes. *Forest
757 Policy and Economics* 13(6): 425–434.
- 758 Pukkala, T. 2014. Does biofuel harvesting and continuous cover management increase carbon
759 sequestration? *Forest Policy and Economics* 43: 41–50.
- 760 Pukkala, T. 2017. Does management improve the carbon balance of forestry? *Forestry: An
761 International Journal of Forest Research* 90(1): 125–135.
- 762 Repola, J. 2009. Biomass equations for Scots pine and Norway spruce in Finland. *Silva Fennica
763* 43(4): 625–647.
- 764 Reyer, C.P.O., S. Bathgate, K. Blennow, J.G. Borges, H. Bugmann, S. Delzon, S.P. Faias, J. Garcia-
765 Gonzalo, B. Gardiner, and J.R. Gonzalez-Olabarria. 2017. Are forest disturbances amplifying
766 or canceling out climate change-induced productivity changes in European forests?
767 *Environmental Research Letters* 12(3): 34027.
- 768 Ruosteenoja, K., K. Jylhä, and M. Kämäräinen. 2016. Climate Projections for Finland Under the
769 RCP Forcing Scenarios. *Geophysica* 51.
- 770 Sathre, R. and J. O'Connor. 2010. Meta-analysis of greenhouse gas displacement factors of wood

771 product substitution. *Environmental Science & Policy* 13(2): 104–114.

772 Seidl, R., D. Thom, M. Kautz, D. Martin-Benito, M. Peltoniemi, G. Vacchiano, J. Wild, D. Ascoli,
773 M. Petr, and J. Honkaniemi. 2017. Forest disturbances under climate change. *Nature Climate*
774 *Change* 7(6): 395.

775 Seppälä, J., T. Heinonen, T. Pukkala, A. Kilpeläinen, T. Mattila, T. Myllyviita, A. Asikainen, and
776 H. Peltola. 2019. Effect of increased wood harvesting and utilization on required greenhouse
777 gas displacement factors of wood-based products and fuels. *Journal of Environmental*
778 *Management* 247: 580–587.

779 Skog, K.E. 2003. Harvested wood products: basis for future methodological development. *Good*
780 *Practice Guidance for Land Use, Land-Use Change and Forestry [Electronic Resource].*
781 *Hayama, Kanagawa, Japan: Published by the Institute for Global Environmental Strategies*
782 *(IGES) for the Intergovernmental Panel on Climate Change (IPCC), C2003:[16] Pa.*

783 Smyth, C., G. Rampley, T.C. Lemprière, O. Schwab, and W.A. Kurz. 2017. Estimating product and
784 energy substitution benefits in national-scale mitigation analyses for Canada. *Gcb Bioenergy*
785 9(6): 1071–1084.

786 Soimakallio, S., R. Antikainen, and R. Thun. 2009. Assessing the sustainability of liquid biofuels
787 from evolving technologies. *A Finnish Approach. Technical Research Centre of Finland (VTT)*
788 *Research Notes* 2482.

789 Soimakallio, S., L. Saikku, L. Valsta, and K. Pingoud. 2016. Climate Change Mitigation Challenge
790 for Wood Utilization □ The Case of Finland. *Environmental Science & Technology* 50(10):
791 5127–5134.

792 Statistics Finland. 2010. *Greenhouse Gas Emissions in Finland. National Inventory Report under*
793 *the UNFCCC and the Kyoto Protocol. 25 May 2010.*

794 Suter, F., B. Steubing, and S. Hellweg. 2017. Life cycle impacts and benefits of wood along the
795 Value chain: the case of Switzerland. *Journal of Industrial Ecology* 21(4): 874–886.

- 796 Tuomi, M., R. Laiho, A. Repo, and J. Liski. 2011a. Wood decomposition model for boreal forests.
797 *Ecological Modelling* 222(3): 709–718.
- 798 Tuomi, M., J. Rasinmäki, A. Repo, P. Vanhala, and J. Liski. 2011b. Soil carbon model Yasso07
799 graphical user interface. *Environmental Modelling & Software* 26(11): 1358–1362.
- 800 UN. 2015. *Paris agreement*. United Nations, 25 p.
- 801 UNECE/FAO. 2010. *Forest Product Conversion Factors for the UNECE Region*. Geneva,
802 Switzerland: United Nations Economic Commission for Europe, Timber Section, Food and
803 Agriculture Organization for the United Nations.
- 804 Vares, S., T. Häkkinen, and T. Vainio. 2017. *Rakentamisen hiilivarasto*.
805 [http://www.ym.fi/download/noname/%7B2859F537-ECD2-479D-A62B-](http://www.ym.fi/download/noname/%7B2859F537-ECD2-479D-A62B-F13AD75403F2%7D/136827)
806 [F13AD75403F2%7D/136827](http://www.ym.fi/download/noname/%7B2859F537-ECD2-479D-A62B-F13AD75403F2%7D/136827).
- 807 Zubizarreta-Gerendiain, A., T. Pukkala, and H. Peltola. 2016. Effects of wood harvesting and
808 utilisation policies on the carbon balance of forestry under changing climate: a Finnish case
809 study. *Forest Policy and Economics* 62: 168–176.
- 810