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1 Forest bioenergy harvesting changes carbon balance and risks biodiversity in boreal forest
2 landscapes

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12 requirements

13

15 **Abstract**

16 Climate solutions relying on forest bioenergy may be in conflict with carbon sequestration and
17 storage by forests as well as conservation of biodiversity. We quantified effects of forest residue
18 harvesting for bioenergy on both forest carbon balance and biodiversity in a boreal forest
19 landscape. Through a modeling framework we simulated forest development in four real
20 watersheds with three scenarios: i) with and ii) without forest residue harvesting, and iii) set
21 aside to study the conservation potential of these landscapes in the future without management.
22 We simulated changes in the forest carbon stocks, and in the quality and the quantity of
23 deadwood resources for 100 years and combined this information with the information on
24 species habitat associations based on expert judgements. In this study current practices of slash
25 and stump harvesting reduced forest carbon stocks and deadwood volumes at the landscape
26 scale, and consequently halved the emissions savings that can be obtained with bioenergy. In
27 addition, logging residue harvesting reduced 15-21% the combined species conservation capacity
28 of the landscape for red-listed, saproxylic species compared to forest management without
29 bioenergy harvesting. Furthermore, the results indicated a potential conflict between areas of
30 high bioenergy potential and high conservation potential.

32

33 **Introduction**

34 Climate solutions relying on forest bioenergy may be in conflict with carbon sequestration and
35 storage by forests as well as conservation of biodiversity. Intensifying biomass harvests for
36 bioenergy production may cause significant net losses of carbon from forests which may partly
37 or entirely offset the emission savings from replacing fossil fuels with bioenergy (Schulze et al.
38 2012). In addition, the intensification of forest biomass harvests to meet the climate goals has
39 raised concerns on adverse effects on forest biodiversity (EASAC 2017).

40 Logging residues, such as branches, treetops, and stumps, are an increasingly important source of
41 bioenergy in northern temperate and boreal forests. While biomass from other side streams and
42 waste from forest industry are already used for energy production (Szabó et al. 2011), many
43 studies have identified a large, unused bioenergy potential of logging residues (e.g. de Wit and
44 Faaij 2010). As a result of policies promoting bioenergy and concerns about climate change,
45 extraction rates of these previously unharvested residues are expected to further increase in the
46 future (Mantau et al. 2010). This development can already be seen in Sweden (de Jong and
47 Dahlberg 2017) and in Finland (Peltola 2014). In Finland, bioenergy production from forest
48 chips made mainly from logging residues has multiplied by eight times since the year 2000, and
49 placed forest chips as the most important solid wood fuel in Finland (Peltola 2014). Logging
50 residues are an attractive source of bioenergy because the use of residues does not involve a
51 change in land use nor cause a direct competition for land with food production. Hence, unlike

52 the use of agricultural biomass, the use of logging residues is not capped in the EU Renewable
53 Energy directive and use of logging residues will likely expand (COM 2016).

54 Large-scale logging residue harvesting may pose a conflict with conservation efforts to protect
55 deadwood-dependent species (Bouget et al. 2012, Ranius et al. 2018). In Fennoscandia
56 approximately one quarter of forest species depend on deadwood (Siitonen 2001).
57 Approximately half of the threatened forest species in Finland require forests rich in decaying
58 wood (Hyvärinen et al. 2019). While national initiatives have been set to preserve and increase
59 the amount of deadwood in forests to improve the state of biodiversity (Hjältén et al. 2010),
60 simultaneous large-scale forest residue harvesting may result in further losses in already scarce
61 deadwood resources and additional detrimental effects on deadwood-dependent species (e.g.
62 Ranius et al. 2014, Johansson et al. 2016).

63 Previous studies indicate that logging residue harvesting for bioenergy causes habitat reduction
64 and destruction and changes in the temporal availability of habitats of deadwood-dependent
65 species, but the long-term impacts remain uninvestigated (Ranius et al. 2018). Previous empirical
66 studies investigating the effects of forest bioenergy on biodiversity have focused mainly on
67 short-term effects at the stand-level (de Jong & Dahlberg, 2017) or provided a snapshot field data
68 at landscape level (Hiron et al. 2018). Modeling approaches have been introduced to investigate
69 longer term impacts of logging residue extraction on deadwood (Verkerk et al. 2014; Hof et al.
70 2018a) or deadwood-dependent species (Geijer et al. 2014; Ranius et al. 2014; Johansson et al.
71 2016; Snäll et al. 2017). These modelling studies have either considered single forest stands or
72 theoretical or real landscapes, and as response variables they have used the amount of dead

73 wood, the population development of real or theoretical model species, or the habitat amount for
74 a wide range of saproxylic organisms based on their specialization to deadwood types. Studies
75 investigating deadwood dynamics or changes in different deadwood types over time on larger
76 landscapes with variation in initial, measured conditions are scarce. The information on
77 deadwood types is important when assessing long-term impacts of logging residue harvesting,
78 because both volume and diversity of deadwood affect the long-term sustainability of
79 populations of deadwood-dependent species (Tikkanen et al. 2007). Hence, despite the large
80 body of empirical research, little is still known about how forest biodiversity would respond to a
81 large-scale removal of logging residues in the long term, and at the landscape scale. The lack of
82 long-term studies limit our ability to predict the effects of harvesting logging residues on
83 deadwood in two ways. First, only long-term studies are sufficient to capture the relatively slow
84 process of wood decay dynamics. Second, year to year fluctuations in the deadwood diversity
85 and abundance are common, and these fluctuations may mask the effects in short-term field
86 experiments (Riffell et al. 2011).

87 The question whether bioenergy from logging residues truly mitigates climate change has been
88 substantially debated both in public (Kangas et al. 2018) and in scientific forums (Agostini et al.
89 2013). An increasing number of studies show that logging residue extraction decreases the
90 carbon stock and the sink capacity of forests, which reduces the climate benefits of replacing
91 fossil fuels with bioenergy for years and decades (Agostini et al. 2013). However studies based
92 on single stands may not reflect carbon dynamics in larger landscapes. While the carbon loss due
93 to logging residue removal on individual stands is acknowledged, it may have not been
94 considered problematic because other stands in the landscape could act as carbon sinks

95 compensating the carbon loss (Lamers and Junginger 2013). The initial age structure of the forest
96 stands affects the carbon balance of forest landscape (e.g. Routa et al. 2012). Hence, analyses on
97 larger landscapes are needed to investigate the effects of large-scale logging residue harvesting
98 on carbon dynamics.

99 A central question in the discussion on the effects of logging residue extraction is if landscape
100 level processes could counteract the adverse effects of residue extraction on carbon balance or
101 deadwood. Hence, even in landscapes where large-scale logging residue harvesting takes place,
102 different types of deadwood could be abundant or increasing in other parts of the landscape
103 where logging residue harvesting is not practiced (Lamers and Junginger 2013). This implies that
104 logging residue harvesting would not decrease deadwood availability for saproxylic species at
105 the landscape level. Similar arguments have been presented for forest carbon balance (Lamers
106 and Junginger 2013). Yet, studies quantifying both changes in forest carbon balance resulting
107 from logging harvesting and subsequent effects on the habitat availability of deadwood-
108 dependent species on larger landscapes are missing. In addition, it has not been investigated to
109 what extent forest stands that have a high bioenergy potential also have a high conservation
110 potential for red-listed, deadwood-dependent species. Evaluating this potential overlap in forest
111 values is important from the forest planning perspective.

112 The objectives of this study were to i) investigate how large-scale stump and slash harvesting for
113 bioenergy affects forest carbon balance and the quality and quantity of deadwood in a landscape,
114 ii) study how the changes in the deadwood affect the amount of suitable habitats of deadwood-
115 dependent species and iii) explore the overlap between areas of high bioenergy potential and
116 high conservation potential for selected deadwood-dependent species. To address these questions,

117 we modeled the forest carbon dynamics and the availability of different types of deadwood in a
118 landscape over 100 years.

119 **Material and Methods**

120 **Approach**

121 To study the effect of large-scale forest residue harvesting on forest carbon balance and the
122 habitat availability on deadwood-dependent species we simulated the development of forest
123 stands in central Finland. Our study landscape consisted of four watersheds that differed in their
124 productivity and age distribution (Table 1). The modelled landscape encompassed of ca. 11 000
125 hectares and was derived from 6 800 forest stands on mineral soils (Table 1). Measured forest
126 data was provided by the Finnish Forest Center. Our simulation combined modelling of forest
127 carbon budget, deadwood quality and quantity with information of species habitat associations.
128 Species habitat associations modelled with habitat suitability indices were based on expert
129 judgements (Tikkanen et al. 2006, 2007). Combining expert judgements with forest models has
130 been used also in previous studies, but on individual stands (Dahlberg et al. 2011; Ranius et al.
131 2014). In addition, instead of a snapshot of time, our approach provided dynamic estimates of
132 bioenergy resource, forest carbon balance, and the formation and the decay of different types of
133 deadwood in larger forest landscapes

134

135 We simulated three scenarios i) with (BIO) and ii) without (BAU) forest residue harvesting for
136 bioenergy, and iii) set aside (SA), which was used to study the conservation potential of these
137 landscapes without management. In the BIO and BAU scenarios forest stands were managed
138 according to the current forest management recommendations in Finland where forests were
139 clear-cut at the age of 70-90 years (Äijälä et al. 2014). After final felling stands were artificially
140 regenerated by planting or seeding. Stands were tended and thinned two to three times before the
141 final felling. In the BIO scenario branches and treetops were harvested from thinnings and final
142 fellings. In addition, stumps were extracted from all clear-cuts. We assumed that 70% of all
143 available residues were harvested, which was consistent with the extraction percentage reported
144 in field studies (Dahlberg et al. 2011), and with good practice guidance for forest residue
145 harvesting in Finland (Koistinen et al. 2016). We assumed that logging residue harvesting
146 increased the destruction and removal of snags and downed logs by 30% based on the
147 observations of field studies (Hautala et al. 2004; Rudolphi and Gustafsson 2005; Rabinowitsch-
148 Jokinen and Vanha-Majamaa 2010). To define the increase in destruction and removal in the
149 modelling framework we used an iterative approach where we first estimated the baseline
150 reduction in the annual deadwood input to be 60% in the BAU scenario. The rate of destruction
151 and removal resulted deadwood pool values consistent with the measured values in the National
152 Forest Inventory (Peltola 2014). High rate of deadwood destruction and removal are partly
153 explained collection of firewood by forest owners. Foliage was assumed to be left in stands to
154 prevent nutrient loss and corrosion in power plants (Alakangas et al. 2016). We assumed no
155 effect on the growth of the next tree generation.

156 We predicted the forest growth and yield for each stand with a SIMO forest simulator for 100
157 years. The SIMO modelling framework consists of several models for describing natural
158 processes, such as growth and mortality, and forestry operations. These models are documented
159 in detail in the scientific literature (Kangas and Rasinmäki 2008). The growth and yield
160 predictions were made using growth models by Hynynen et al. (2002), which were developed
161 based on extensive data of field measurements in the National Forest Inventory. Therefore, the
162 models based on this data cover all main tree species and forest site types in Finland (Hynynen et
163 al. 2002). To predict the development of total biomass, simulated stem wood volumes were
164 converted to estimates of total aboveground and belowground biomass through biomass models
165 by Repola et al. (2007) within the SIMO framework. The SIMO model has been shown to
166 produce equally good estimates of forest growth as another widely used Finnish forest simulator
167 MOTTI (Mäkinen et al. 2008).

168 To estimate changes in litter and soil carbon stocks resulting from forest harvest residue
169 extraction, the litter input from SIMO was used as input to the soil carbon model Yasso07
170 (Tuomi et al. 2011). Litter input to soil included input from thinnings and final fellings (Kangas
171 and Rasinmäki 2008), natural mortality (Hynynen et al. 2002), and litter input from living trees
172 (Liski et al. 2002). In addition, litter input from understory vegetation to soil was accounted for
173 (Muukkonen and Mäkipää 2006). In the Yasso07 model the decomposition of organic matter
174 depends on climate, litter type and litter diameter (Tuomi et al. 2011). We applied 2 cm diameter
175 for all fine woody litter and an average diameter of 10 cm for coarse woody litter (Raumonen et
176 al. 2011). The Yasso07 model calculates carbon stocks separately for each input type and the
177 sum of these carbon stocks is the total litter and soil carbon stock. We used the carbon stock,

178 calculated from the coarse woody litter without the humus fraction, as a proxy for coarse woody
179 debris.

180 The initial litter and soil carbon stocks were calculated by running the Yasso07 model to a steady
181 state with an average litter input of current recommended rotation periods (Äijälä et al. 2014). To
182 account for the different stand ages in the landscape in the initialization, the simulated steady-
183 state carbon stocks were adjusted to match the site type and initial age. For this, soil carbon pools
184 were estimated for different stand ages using the information about annual litter input of
185 recommended rotation period. These values were assigned for stands based on stand age, site
186 type and dominant species.

187 We estimated the potential of the forest landscape consisting of four watersheds to provide net
188 CO₂ emissions reductions with bioenergy while taking into account the changes in forest carbon
189 balance. Bioenergy potentials was calculated by applying net calorific values of 19.3 MJ kg dry
190 biomass⁻¹ for stumps and 19.6 MJ kg dry biomass⁻¹ for branches and treetops (Alakangas et al.
191 2016). The net caloric values were averages of different deciduous and coniferous species. The
192 effect of bioenergy harvesting on the forest carbon balance was the difference between BAU and
193 BIO scenarios. To show the magnitude of possible net emission reductions with bioenergy from
194 forest harvest residues we applied the emission factor of 93 kg CO₂ GJ⁻¹ to coal (Statistics
195 Finland 2019).

196 **Modelling the changes in the available habitat for red-listed deadwood-dependent species**

197 To simulate changes in the amount and the quality of snags and downed logs we linked
198 predictions of the natural mortality (Hynynen et al. 2002) and decomposition through decay
199 stages (1-5) of deadwood (Mäkinen et al. 2006). These models predict 1) initial wood density of
200 the dead tree, 2) probability of dead tree being a snag or a downed log and 3) the change to
201 successive decay class. Initial values for snags and downed deadwood were set to be consistent
202 with the Finnish National Forest inventory data, which provided average volumes per hectare of
203 snags and downed deadwood for tree species for Southern Finland (Peltola 2014). The initial
204 values for snags and downed logs together ranged from 3.1 to 4.5 m³ ha⁻¹, and the values were
205 assigned as initial values based on the dominant tree species on studied stands (Cajander 1949).

206 To study the effects of the scenarios on the availability of the habitats of selected red-listed
207 deadwood dependent species we applied models for habitat suitability indices (HSIs) to calculate
208 species conservation capacity (SCC). This resource-focused approach was chosen as the
209 reduction of deadwood along with reduction of old-growth forests and decreasing number of
210 large trees is the primary cause of threat to threatened forest species, and the second most import
211 cause of regional extinctions in Finland (Hyvärinen et al. 2019). We applied HSI models that
212 connect forest stand characteristics with habitat requirements of red-listed species. The HSI
213 models developed by Tikkanen et al. (2006, 2007) have been created using expert opinion
214 assessment in which Finnish experts in various taxa assessed the habitat preferences of the red-
215 listed forest species in relation to predefined structures of forest stands and trees within stands.
216 Then a total of 98 species were grouped to 27 groups according to their shared habitat
217 requirements, which can be connected to stand characteristics modelled with a forest simulator

218 (Tikkanen et al. 2007). Each group is represented by one red-listed species that typifies the group
 219 (Tikkanen et al. 2007). The type species cover 11 fungi, 15 insect and one lichen species. The
 220 stand specific HSI of a type species is a product of deadwood resources, microclimate and
 221 temporal continuity of the resource (see Tikkanen et al. 2007 for equations). We estimated the
 222 combined SCC in different scenarios for each stand s for different HSIs describing stand quality
 223 for type species k (eq 1, Pakkala et al. 2002). The combined SCC is the weighted average of
 224 HSIs, in which the HSIs give the weights. The combined SCC was used as a proxy for a
 225 conservation capacity of a stand for all 27 type species for each year. Then the average combined
 226 SCC across landscape for each year was calculated for BAU, BIO and SA to compare scenarios.
 227 In addition, to study the effect of residue harvesting on HSIs of different type species, we
 228 calculated type species-specific vulnerability to residue harvesting with the methodology
 229 introduced by Mazziotta et al. (2016). Type species-specific SCC were calculated to form a
 230 proxy for the conservation capacity of all stands for each type species and year in BIO and BAU
 231 scenarios. Vulnerabilities were calculated as the differences in the species-specific SCC across
 232 stands between BIO and BAU scenarios for each year. These vulnerabilities were summarized
 233 across landscape and time for different combinations of species requirements for deadwood
 234 resource and microclimate.

$$235 \quad SCC_s = \frac{\sum_{k=1}^{27} ((HSI_s)_k)^2}{\sum_{k=1}^{27} (HSI_s)_k} \quad (1)$$

236

237 **Relationships between areas of high bioenergy potential and high potential conservation**
 238 **capacity**

239 We identified stands with high bioenergy potential and high conservation capacity by classifying
240 stands by combined SCC values in the set aside scenario, and by bioenergy potential in the
241 bioenergy scenario to top 10 % quantiles. This threshold was chosen because the 10% quantiles
242 have been used to identify important sites for conservation planning and ecosystem service
243 hotspots (Schröter et al. 2017). The stands were classified to the highest class if the value was
244 over the highest-class threshold at any point of the simulation period.

245 **Results**

246 **Changes in forest carbon stocks and balance**

247 Forest residue harvesting for bioenergy reduced litter and soil carbon stocks on average 3.5 tC ha
248 ⁻¹ the beginning of the simulation period and up to 12 tC ha ⁻¹ after 100 years. This carbon loss
249 corresponded to a 3-9% decrease in the size of litter and soil carbon stock. Logging residue
250 harvesting reduced the average amount of coarse woody debris 15-31% per hectare.
251 Development of total carbon stocks varied across watersheds in response to initial age
252 distribution and site productivity. Nevertheless, the reduction in the litter and soil carbon stocks
253 resulting from residue harvesting was of the same order of magnitude in all watersheds, hence
254 the results were presented for all four watersheds together.

255 Producing bioenergy from forest residues reduced fossil fuel emissions, but also reduced the
256 forest carbon sink compared to scenario where residues were left in place (Figure 1). The annual
257 average bioenergy potential to replace fossil fuels in the landscape of four watersheds ranged
258 from 323 to 1100 TJ year⁻¹ depending on timber harvests in different years. Harvesting stumps,
259 branches and treetops for bioenergy reduced the carbon sink of the landscape (Figure 1, white

260 bars). Although bioenergy reduced fossil fuel emissions (Figure 1 grey bars), the decrease in the
261 carbon sink reduced the net emission savings. Since the logging residues would release CO₂ even
262 if left to decompose in the forest, the negative effect on the forest sink decreased over time.
263 Hence, after ten years the decrease in the carbon sink reduced the net emission savings 53%,
264 12% after 50 years and 2% after 100 years (Figure 1, grey and black bars).

265 **Changes in deadwood and the habitat availability for deadwood-depended species**

266 The destruction and removal of snags and logs resulting from logging residue extraction reduced
267 the average amount of deadwood in the landscape 1 – 3.7 m³ ha⁻¹ during the simulation period,
268 and changed the quality of deadwood, especially reducing the amount of recently killed logs or
269 snags of coniferous species (Figure 2) compared to management without residue harvesting. The
270 average relative reduction in the BIO scenario compared to BAU was the largest in recently died
271 spruce logs and snags (-43, and -41%), recently died deciduous logs (-38%) and the smallest in
272 almost totally decayed pine logs (-19%) during the study period. Residue harvesting caused
273 bottlenecks in the availability of at least one deadwood type (volume < 0.001 m³ ha⁻¹) on 185–
274 2724 forest stands, which corresponds to 0.07 – 1.7% increase in bottlenecks compared to the
275 BAU scenario. Deadwood was generally a scarce resource in the simulated landscapes with the
276 average total amount of deadwood ranging from 2.2 to 10.1 m³ ha⁻¹ in the BAU scenario. In the
277 set aside scenario, average total amount of deadwood in the landscape was 6–73 m³ ha⁻¹. Hence,
278 the set aside landscape had 2.5 – 33 times more deadwood per hectare than the same landscapes
279 managed according to the current recommendations. Logging residue harvesting further
280 increased this difference.

281 Logging residue extraction reduced the combined species conservation capacity of the landscape
282 for the selected deadwood-dependent type species on average by 15 – 21% compared to BAU and
283 32-54% compared to the set aside scenario (Figure 3). The species-specific vulnerabilities to
284 residue harvesting were up to 8% during the simulation period. No large differences in the
285 vulnerability to residue harvesting were observed between different type species (Figure 4).
286 Consequently, irrespective of the specific resource or microclimatic associations of the species,
287 bioenergy harvesting reduced the ability of the landscape to provide habitats for the studied red-
288 listed species.

289 **Relationships between high bioenergy and conservation potential**

290 In the data 17% of stands had both the greatest conservation capacity and the greatest potential
291 for bioenergy exploitation when threshold for the highest potentials were set to top 10% quantile
292 (Figure 5). However, 31% of the forest stands had high bioenergy potential but low conservation
293 potential. Herb-rich heath forests, with Norway spruce as the dominant species, had both the
294 highest bioenergy and the highest conservation potential. Alternatively these characteristics
295 applied also to stands with the highest bioenergy and the lowest conservation potential (Figure
296 5). Hence, simple forest characteristics such as site type or stand age did not identify stands with
297 high bioenergy potential and high/low conservation potential.

298 **Discussion**

299 In this study logging residue harvesting reduced both forest carbon stock and carbon sink
300 capacity of forest landscapes, and decreased deadwood compared to business as usual
301 management without bioenergy harvesting. Logging residue harvesting reduced the amount of
302 snags and logs in different decay stages at landscape scale faster than these deadwood resources

303 were formed, and the same applied to litter and soil carbon stocks. Hence, large-scale logging
304 residue harvesting increased the outflow of carbon and deadwood from the system compared to
305 business as usual management. Our results show that logging residue harvesting decreases
306 carbon stocks and deadwood availability for saproxylic species even when carbon dynamics are
307 analyzed on landscape level instead of individual stands. These findings corroborate the findings
308 of other studies on stand (e.g. Ranius et al 2011) and landscape level (e.g. Johansson et al. 2016.
309 Hiron et al. 2018)

310 Changes in the forest carbon balance resulting from logging residue harvesting reduced the
311 climate benefits of bioenergy and resulted in delayed emissions savings. The results support the
312 earlier studies on different scales ranging from forest stands (Zanchi et al. 2012) to forest
313 landscapes (McKechnie et al. 2011) and countries (Repo et al. 2015a) showing that bioenergy
314 from logging residues is not carbon or climate neutral, and that emission savings with bioenergy
315 come with a delay. In our study after the first decade of residue harvesting changes in carbon
316 balance reduced the net emissions savings by half compared to fossil carbon savings. Reductions
317 of similar magnitude in emission savings were reported by Forsius et al. (2016) in their study
318 conducted in the same region. During the simulation period, the forest carbon stock of the
319 landscape increased in all scenarios because of initial conditions of the studied landscape.
320 Although initial age distribution and site productivity affected the forest carbon dynamics (Routa
321 et al. 2012), accounting for these factors did not compensate for the reduction in the carbon
322 stocks due to bioenergy harvesting. We assumed that logging residue harvesting does not affect
323 the growth of the next tree generation. Field experiments on whole-tree harvesting report no
324 effect and negative effects of tree growth after final felling (Thiffault et al. 2011) and growth

325 losses after repeated whole-tree harvesting (Kaarakka et al. 2014). If forest residue harvesting
326 caused growth loss, the effects on carbon balance would be more pronounced.

327 Logging residue harvesting, and the associated destruction of snags and downed logs, halved the
328 amount of some decay classes of logs and snags, and reduced the amount of coarse woody debris
329 for up to one third. Field studies have reported an additional damage, or an immediate loss of
330 deadwood, due to logging residue harvesting ranging from 25 to 88% depending on the
331 definition of deadwood (Eräjää et al. 2010; Rabinowitsch-Jokinen and Vanha-Majamaa 2010).
332 Some field studies have highlighted the destruction of moderately and well-decayed coarse
333 woody debris (Rabinowitsch-Jokinen and Vanha-Majamaa 2010, Work et al. 2014), while in the
334 current study the largest relative reduction was in the fresh deadwood. In general, in our study
335 the average reduction in the coarse woody debris was lower than reported in field studies. The
336 use of different definitions for deadwood and decay classes make direct comparisons to other
337 studies challenging.

338 Logging residue harvesting and the associated destruction of snags and downed deadwood
339 reduced the combined species conservation capacity of the landscape for deadwood-dependent
340 red-listed species by one fifth compared to the general effect of forest management. The result
341 may be an under- or overestimation because of limited empirical data on the on additional effect
342 of residue harvesting on destruction and removal of large-diameter deadwood. However, our
343 results highlight the importance of considering this potential loss of large-diameter deadwood in
344 bioenergy harvesting operations. In Finland and Sweden, the majority of declining and red-listed
345 deadwood-dependent species are associated with coarse woody debris rather than with fine

346 woody debris (Siitonen 2001). As none of the red-listed species in Sweden is primarily
347 associated with branches and tree tops of Norway spruce, a review study by de Jong and
348 Dahlberg (2017) suggested that harvesting logging residues of conifers has small to negligible
349 additional impacts on species of conservation interests in Sweden with current extraction levels.
350 However, our study suggests that the long-term impacts of extensive residue harvesting, and the
351 associated removal and destruction of large-diameter deadwood are important. Since the
352 investigated species are largely red-listed because of the reduction in deadwood (Hyvärinen et al.
353 2019), a further reduction of suitable habitats because of bioenergy harvesting operations may
354 have detrimental effects on these species.

355 Our study indicates that combating climate change with bioenergy makes halting biodiversity
356 loss increasingly difficult. In a study by Snäll et al. (2017), stump harvesting for bioenergy
357 caused a rapid decline in lichen metapopulations rendering many currently common lichen
358 species red-listed within two to three decades in Sweden. However, since the lichen
359 metapopulations stabilized at lower equilibrium levels of the stump resource after few decades
360 and climate benefits of stump bioenergy increased with time, Snäll et al. (2017) argued that the
361 trade-off between biodiversity conservation and climate change mitigation with stump bioenergy
362 is transient. This conclusion may result from the assuming a sudden increase in stump harvesting
363 from zero to 50% (Snäll et al. 2017). A slower increase in harvest levels might result in a
364 different conclusion. However, concluding that the trade-off is transient would mean accepting
365 permanently lower population levels for common species. Even if somewhat lower population
366 size does not cause high risk to the common species themselves, it may potentially cause losing
367 some of the specialist species that are dependent on the presence of the common ones. Many

368 saproxylic species are, for example, dependent on the so-called priority effects, where a
369 preceding species changes the physiochemical conditions of the deadwood suitable for successor
370 species (Weslien et al. 2011). It has been shown that the successor suffers more of the reduced
371 availability of the resources (Abrego et al. 2017).

372 Moreover, permanently lower resource availability may be detrimental for species that are
373 already red-listed or depend on specific deadwood types. Our study indicates that logging residue
374 harvesting, and the associated additional deadwood destruction may increase temporal breaks in
375 the availability of the deadwood resources at the landscape scale. These breaks may make the
376 effects of bioenergy harvesting more severe because deadwood continuity has been shown to
377 affect deadwood-dependent biodiversity (Nordén et al. 2014). If such a break happens on a large
378 enough area, some of the specialist species may be totally lost. Johansson et al. (2016) showed
379 that harvesting stumps on 20% of the clear-cuts of the landscape increases extinction risk of rare
380 specialist species to 50%, and further increasing stump extraction to 30% of the clear-cuts
381 negatively affects common species. Thus, extensive logging residue harvesting not only makes
382 halting biodiversity loss less likely, but also increases the likelihood of extinctions of currently
383 threatened species and may put currently common species under risk of becoming threatened.

384 We assessed the reliability of our results by comparing estimates of different modelling steps to
385 independent data. The changes in the carbon balance due to logging residue harvesting result
386 from changes in the litter and soil carbon pool, and the size of the soil carbon pool is determined
387 by litter input and the decay rate of organic matter decomposition. The annual litter input
388 estimates ranged from 1.4 – 3.9 tC ha⁻¹ year⁻¹, which is consistent with other estimates for

389 Southern Finland based on net primary production (Härkönen et al. 2010), litter measurements
390 (Ukonmaanaho et al. 2008; Ilvesniemi et al. 2009), and with other modelling frameworks (Repo
391 et al. 2015a; Lehtonen et al. 2016). The soil carbon stock estimates were within the range 40–
392 149 tC ha⁻¹ reported for Southern Finland in other studies (Rantakari et al. 2012; Repo et al.
393 2015a; Lehtonen et al. 2016). The annual changes in the litter and soil carbon pool were
394 modelled with a soil carbon model, which produced stock change estimates and uncertainty
395 estimates of similar magnitude as other models or soil measurements (Rantakari et al. 2012;
396 Ortiz et al. 2013). The average logging residue potentials per hectare were consistent with
397 previous modelling and field studies (Eräjää et al. 2010; Repo et al. 2015b). The quantitative
398 estimates of net emission savings that can be obtained with forest bioenergy depend on the
399 diameters of residues harvested, time scales studied, energy content of forest chips and the fossil
400 fuel replaced (Repo et al. 2012; Alakangas et al. 2016). However, this study shows the
401 magnitude and time dynamics of the emission savings at landscape scale when forests are
402 managed and residues extracted following the current recommendations. It is important to note
403 that species conservation capacity was derived from potential availability of suitable habitats
404 produced by model and habitat preferences based on expert judgment and our model predictions
405 would benefit from validation in field. However, the main findings were consistent with results
406 from a field study in Sweden (Hiron et al. 2018)

407 Setting safe thresholds for forest residue harvesting requires answering three questions: 1) Where
408 to extract logging residues? 2) Which residues to harvest? 3) How much to harvest? To
409 minimize negative impacts on biodiversity logging residue harvesting should be targeted in
410 stands with low ecological values and avoided in stands with high ecological values (Bouget et

411 al. 2012). In this study, approximately every sixth stand had both high conservation potential and
412 high bioenergy potential indicating a potential conflict between bioenergy and biodiversity
413 objectives. However, almost one third of stands were characterized by high bioenergy but low
414 conservation potential. Identifying these stands would be a useful way to avoid the conflict.
415 However, our results suggest that stand characteristics, such as the main tree species, stand age
416 and site type alone cannot separate high and low biodiversity potential. Therefore, more precise
417 proxies will be needed for planning sustainable bioenergy harvesting. Regarding which residues
418 to harvest, extracting fast decomposing, small diameter branches for bioenergy has shown to
419 have smaller climate warming impact than producing bioenergy from slowly decaying stumps
420 (Repo et al. 2012). While from the perspective of climate change mitigation focusing residue
421 harvesting on fast decomposing branches might be beneficial, also small diameter branches have
422 been shown to host specialist species (Juutilainen et al. 2014). Generally, de Jong et al. (2014)
423 suggested that extracting branches and tops of Norway spruce is less problematic from
424 biodiversity perspective than harvesting stumps. However, in this study the stumps constituted
425 27–55% of the bioenergy potential. Hence, meeting the increasing bioenergy demand needs with
426 only branches requires more hectares for logging residue extraction, which may cause additional
427 destruction of large diameter deadwood from larger areas. The intensity of residue extraction
428 can be considered at the level of individual forest stands and at the level landscapes (Root and
429 Betts 2016). Root & Betts (2016) argued that the question of intensity of residue extraction
430 should be addressed at the landscape scale, since it is not possible to maintain all biodiversity
431 and ecosystem functions at the stand scale. A Swedish expert group concluded that extracting
432 slash on more than 50% or stumps on more than 10-30% of the harvested stands increases

433 species extinction risk (de Jong et al. 2017), and consequently pointed out a need for landscape
434 level planning of residue extraction.

435 Logging residue harvesting is in conflict with the ongoing efforts to increase the amount of
436 deadwood in forests to ensure the favorable state for biodiversity. In the short-term bioenergy
437 from forest residues does not result in deep emission reductions, and it puts a small, but
438 additional, pressure on species already at risk with current forest management practices. Global
439 studies show that biodiversity is likely suffer, if cropland expansion for bioenergy is a major
440 component of climate change mitigation (Hof et al. 2018b). Our study together with previous
441 studies (e.g. Hiron et al. 2018), indicates that biodiversity may be at risk because of bioenergy
442 even without land-use change when forest land stays as forest land but the harvesting intensity is
443 increased. While guidance to good practices in energy wood harvesting along with forest
444 certification and legislation offer tools to minimize negative effects of forest bioenergy
445 harvesting on biodiversity and forest carbon balance, they are likely insufficient in preventing
446 further losses. Landscape level forest management and residue extraction planning, avoiding
447 additional destruction or removal of large-diameter deadwood, setting aside land for
448 conservation and new schemes for compensating deadwood or soil carbon loss may offer ways
449 forward.

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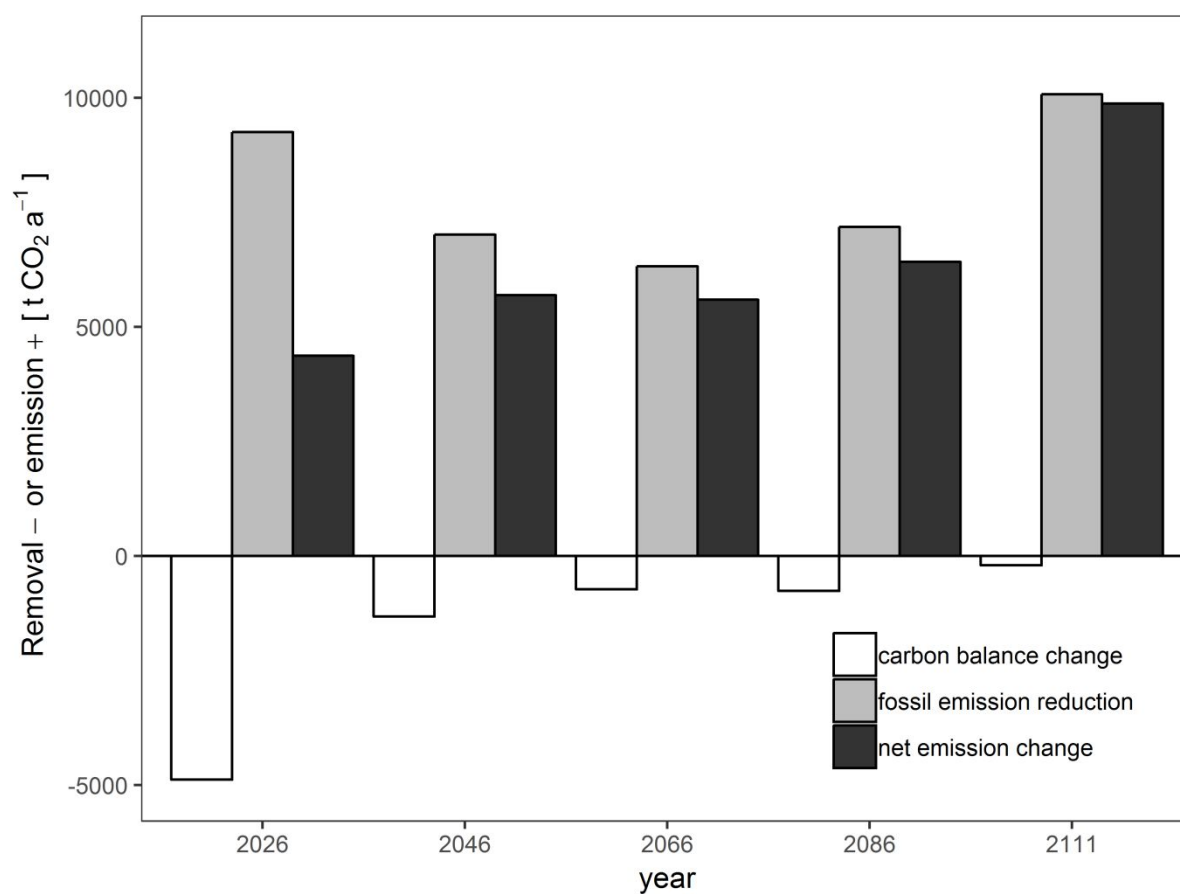
683 **Table 1.** Characteristics of the watersheds. The high productivity site types are classified to herb-
 684 rich heath forest (OMT: see (Cajander 1949) for classification) and mesic heath forests
 685 (MT), while the lower productivity watersheds composed of pine dominated sub-xeric heath
 686 forests (VT) and pine dominated xeric heath forests (CT). The development stage “Young”
 687 refers to a stand with average diameter at breast height of 8-16 cm and “Mature” similarly
 688 average diameter greater than 16 cm but ready for final felling. “Ready to harvest” follows
 689 the timing of final felling according to the current forest management recommendations in
 690 Finland (Äijälä et al., 2014).

691

Site type [% of area]					Development class [% of area]				Stands [nro.]	Total area [ha]
OMT	MT	VT	CT	Clearing	Sapling	Young	Mature	Ready to harvest		
60	38	1	1	4	10	9	39	38	379	473
16	44	38	3	3	19	20	40	19	1142	1766
29	56	15	1	3	18	13	31	35	3258	4709
74	24	1	0	5	18	13	36	29	1923	4073

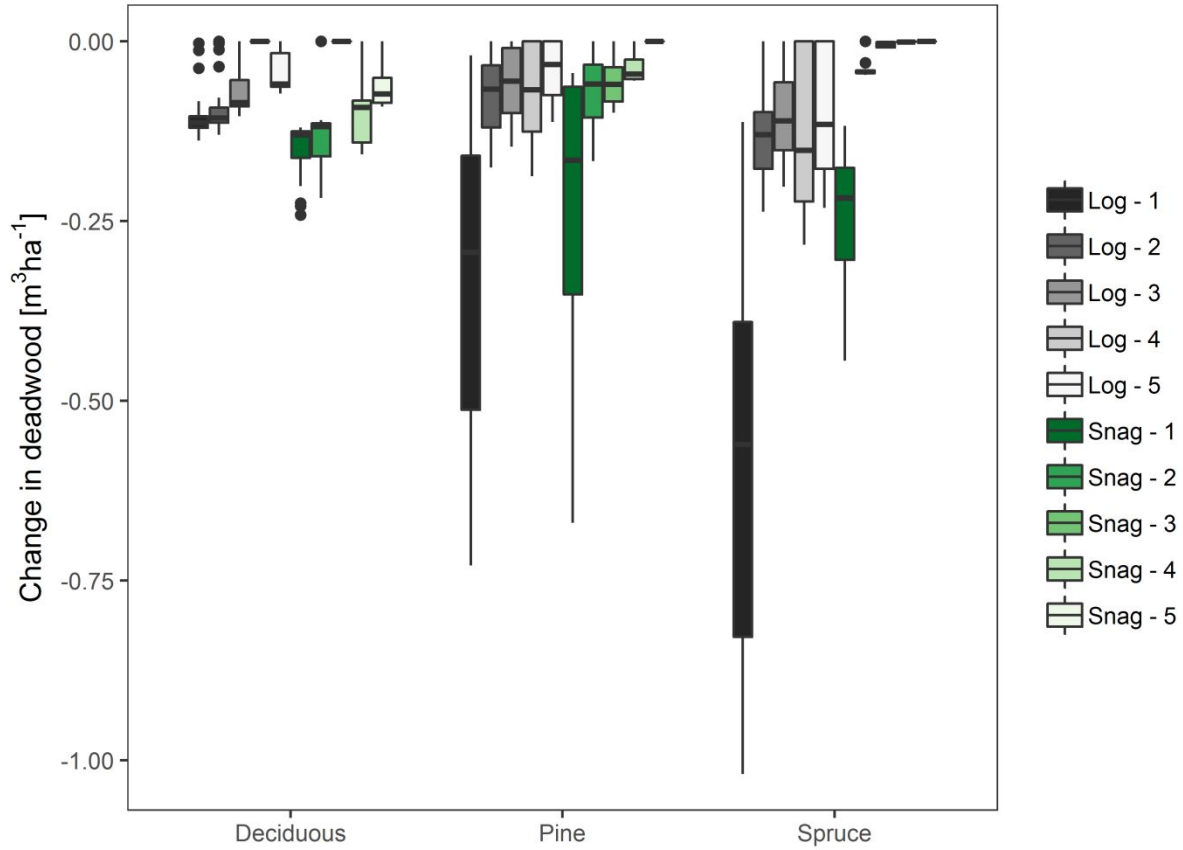
692

694 Figure 1. Changes in fossil emissions, forest carbon balance and net emissions ($\text{t CO}_2 \text{ a}^{-1}$)
 695 resulting from bioenergy production (BIO) compared to no bioenergy harvesting (BAU) in the
 696 landscape consisting of four watersheds. Positive values indicate a change in emissions whereas
 697 negative values a change in removals. Producing bioenergy from forest harvest residues reduces
 698 fossil carbon emissions through substitution (grey) but also cuts the forest carbon sink (white).
 699 The net change is the difference between fossil carbon emission reduction and carbon sink
 700 change (black).



701

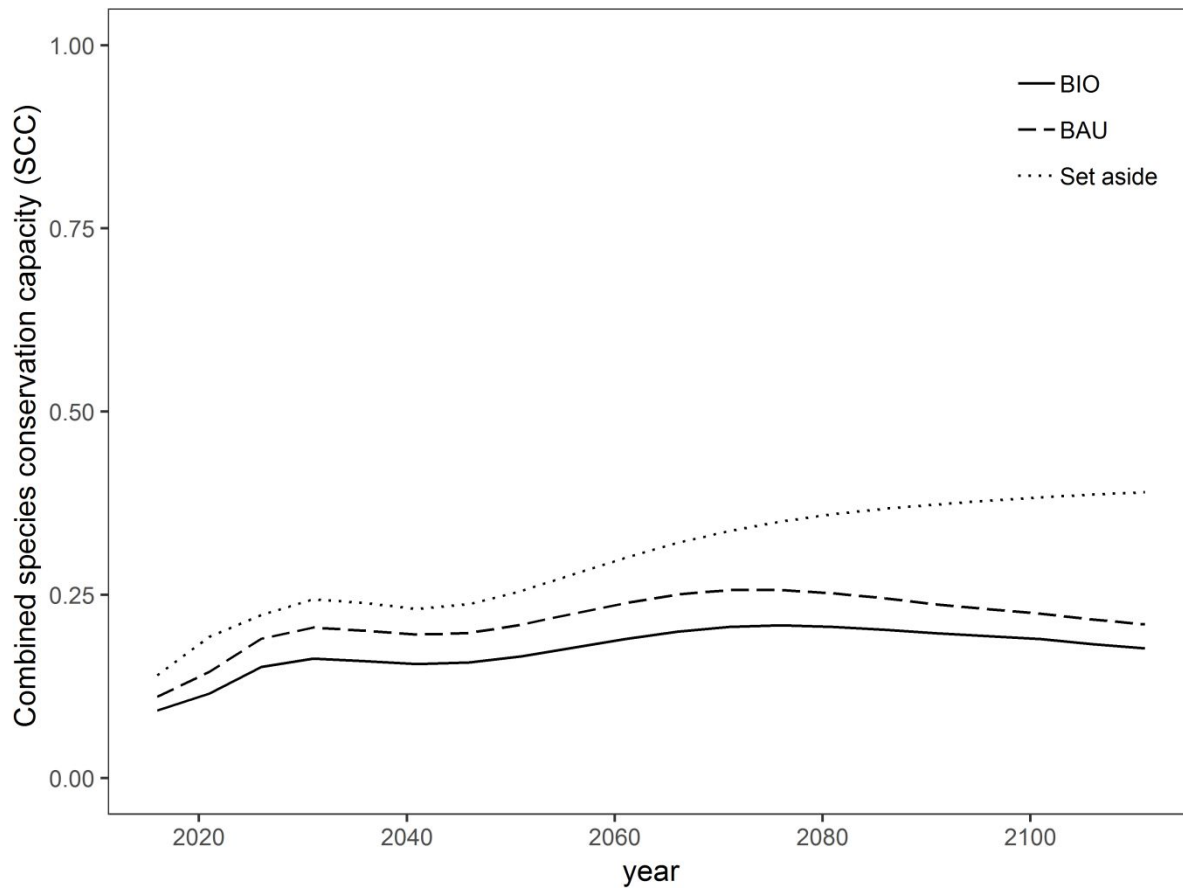
703 Figure 2. Decrease in deadwood types resulting from logging residue harvesting for bioenergy
 704 compared to business as usual management without residue harvesting. Decay stages: 1 =
 705 recently died, 2 = weakly decayed, 3= medium decayed, 4 = very decayed, 5 = almost decayed.



706

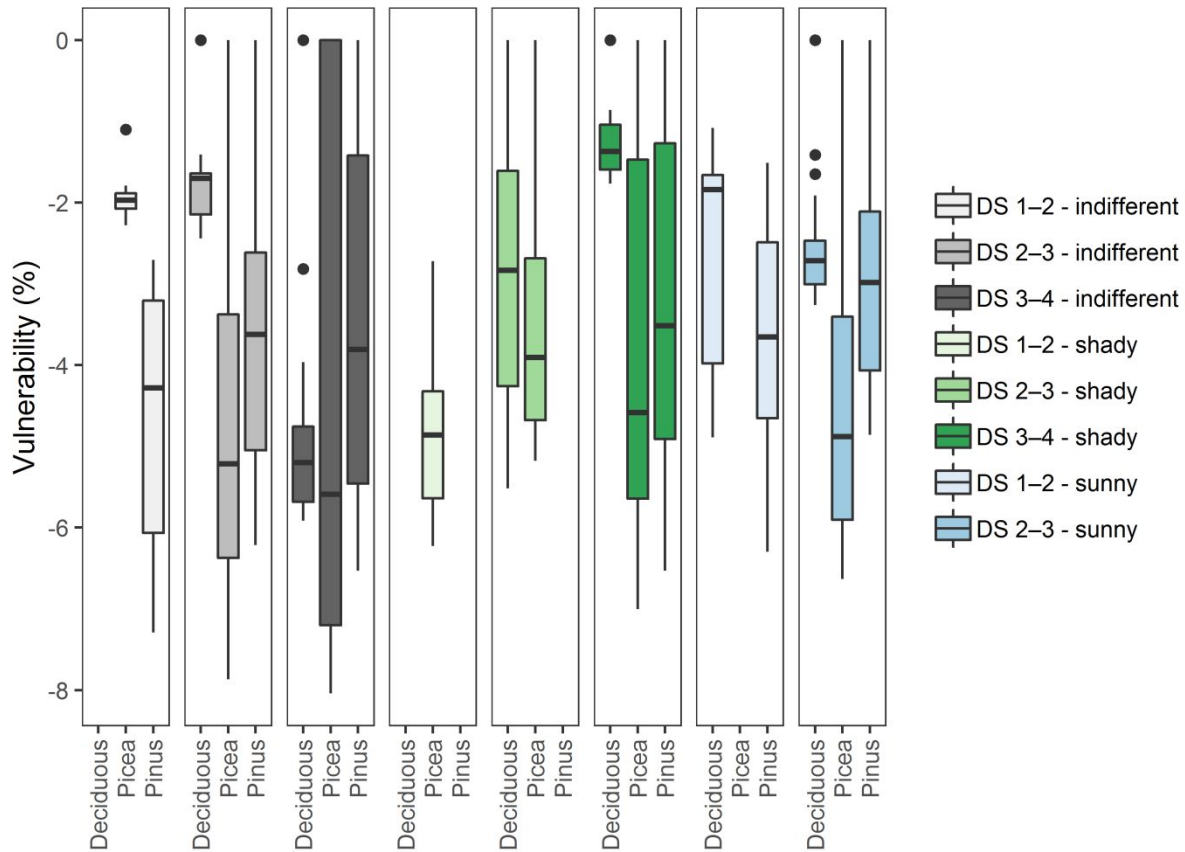
Can. J. For. Res. Downloaded from www.nrcresearchpress.com by JYVASKYLAN YLIOPISTO on 05/28/20
For personal use only. This Just-IN manuscript is the accepted manuscript prior to copy editing and page composition. It may differ from the final official version of record.

707 Figure 3. The development of the average combined species conservation capacity (SCC) in the
 708 landscape in with (BIO) and without (BAU) logging residue residue harvesting for bioenergy,
 709 and without any forest management actions (Set aside) in the landscape.



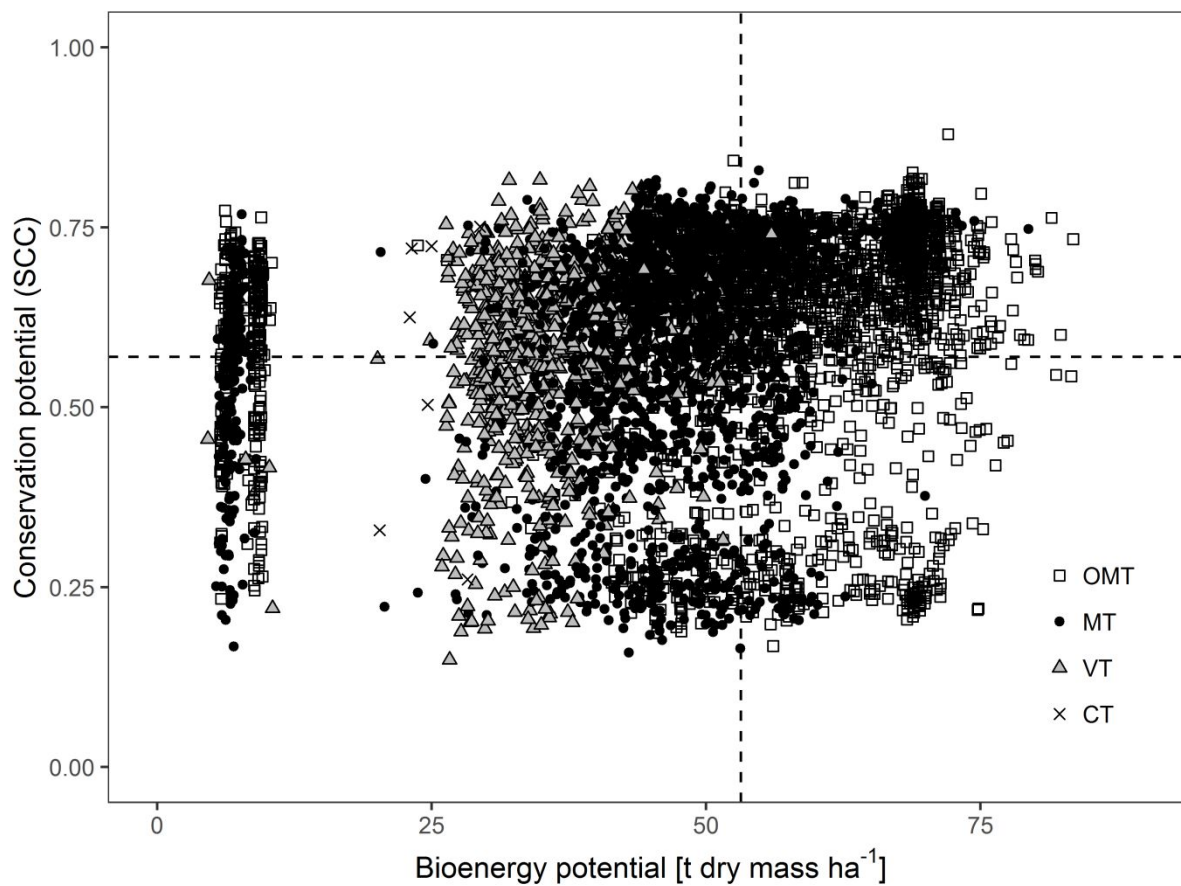
710

711 Figure 4. Type species-specific vulnerability to logging residue harvesting during the simulation
 712 period. The vulnerabilities were calculated as the differences in the type species-specific
 713 conservation capacity across stands between BIO and BAU scenarios for each year. The
 714 vulnerabilities are summarized as percentage values for different combinations of species
 715 requirements for deadwood (DS = decay stage) and microclimate.



716

718 Figure 5. The bioenergy potential and the conservation potential (SCC) of forest stands with
 719 different site types (OMT, MT, VT and CT, see Table 1). The conservation potential is the value
 720 of SCC if stands were set aside in the beginning of the simulation. The dashed lines indicate top
 721 10% quantiles of SCC and bioenergy potential.



722