

This is a self-archived version of an original article. This version may differ from the original in pagination and typographic details.

Author(s): Eyvindson, Kyle; Duflot, Rémi; Triviño, Mária; Blattert, Clemens; Potterf, Mária; Mönkkönen, Mikko

Title: High boreal forest multifunctionality requires continuous cover forestry as a dominant management

Year: 2021

Version: Accepted version (Final draft)

Copyright: © 2020 Elsevier Ltd. All rights reserved.

Rights: CC BY-NC-ND 4.0

Rights url: <https://creativecommons.org/licenses/by-nc-nd/4.0/>

Please cite the original version:

Eyvindson, K., Duflot, R., Triviño, M., Blattert, C., Potterf, M., & Mönkkönen, M. (2021). High boreal forest multifunctionality requires continuous cover forestry as a dominant management. *Land Use Policy*, 100, 104918. <https://doi.org/10.1016/j.landusepol.2020.104918>

1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
21
22

High boreal forest multifunctionality requires continuous cover forestry as a dominant management

Kyle Eyvindson^{1,2,3*}, Rémi Dufлот^{1,2}, María Triviño^{1,2}, Clemens Blattert^{1,2}, Mária Potterf^{1,2}, Mikko Mönkkönen^{1,2}

^{*1}Department of Biological and Environmental Science, University of Jyväskylä, P.O. Box 35, FI-40014 Jyväskylä, Finland, kyle.j.eyvindson@jyu.fi

²School of Resource Wisdom, University of Jyväskylä, P.O. Box 35, FI-40014 Jyväskylä, Finland

³Natural Resource institute Finland (LUKE), Laatokartanonkaari 9, FI-00790 Helsinki

Keywords: Biodiversity; climate change mitigation; continuous cover forestry; ecosystem services; forest planning; optimization.

23 **Abstract:**

24 Intensive extraction of forest resources lowers biodiversity and endanger functioning of forest
25 ecosystems. As such, alternative management regimes have emerged, aspiring to promote forest
26 biodiversity and nature protection in managed forests. Among them, continuous cover forestry, (i.e.
27 selective logging), has received considerable attention and is being promoted by some researchers
28 and NGO's. Yet, the full consequences of banning clear-cuts (i.e. rotation forestry) and replacing it
29 entirely with continuous cover forest remains uncertain. We explore how restricting forest
30 management alternatives (either rotation forestry or continuous cover forestry) will affect
31 landscape-scale forest multifunctionality at a range of harvesting levels. We evaluate
32 multifunctionality as a combination of recreational ecosystem services, climate change mitigation,
33 habitat availability for vertebrates, and red-listed deadwood dependent species. Our results show
34 that restricting forest management alternatives have a negative impact on forest multifunctionality
35 at all harvest levels when compared to the case with no restrictions. Using only continuous cover
36 forestry management alternatives resulted in higher multifunctionality than the case when only
37 rotation forestry management alternatives were used. We also show that maximizing
38 multifunctionality using all management alternatives led to high proportion of continuous cover
39 forestry over the landscape. We conclude that banning clear-cuts does not promote forest
40 biodiversity and multifunctionality at the landscape scale, especially if there is a requirement for
41 high economic benefits required from the forest. However, we recommend that continuous cover
42 forestry should be considered as a primary management alternative, with selective application of
43 rotation forestry wisely planned at the landscape scale.

44 Introduction

45 Biodiversity at a global scale continues to drastically decline even as we improve our understanding
46 of conservation processes (Pimm et al., 2014; Tilman et al., 2014). Increasing human pressure on
47 land-use, the primary driver for terrestrial biodiversity degradation, further hinders conservation
48 efforts (Díaz et al., 2019; Newbold et al., 2015). To reconcile human activities and biodiversity, land-
49 use should be adapted to create multifunctional landscapes that would provide human societies
50 with ecosystem services while maintaining ecosystem integrity. All ecosystems are vulnerable to
51 intensive management; however, some ecosystems seem more resilient than others. Those
52 ecosystems have the largest potential for sustainable resource extraction (Rist et al., 2014). Forests
53 ecosystems have been long time shaped by natural disturbances at different spatio-temporal scales.
54 Therefore, forests should be resilient to resource extraction if managed and viewed at the landscape
55 scale, applying the most efficient silvicultural practices available at the right extent, scale, and
56 intensity (Messier et al., 2019). Forests are of major global interest as a large part of the world's
57 biodiversity relies on forest ecosystems, and they provide a wide range of ecosystem services to
58 human societies, such as timber, water purification, carbon sequestration for climate mitigation or
59 recreational areas (Harrison et al., 2010).

60 Boreal forests, representing approximately one-third of remaining global forests, provide many
61 important ecosystem services (Hansen et al., 2010). Until now, most of boreal forests have been
62 largely preserved from human activities and shelter a large proportion of the remaining wilderness
63 areas at global scale (Watson et al., 2016). However, European boreal forests have been intensively
64 managed over multiple centuries, with accelerated extraction over past decades to provide energy
65 and raw material for saw and pulp mills (Mönkkönen et al., 2018). Yet, managing boreal forests for
66 timber resources conflicts both with provisioning of non-timber ecosystem services (ESS) and
67 biodiversity (BD) conservation (Eyvindson et al., 2018; Pohjanmies et al., 2017; Schwenk et al., 2012;
68 Triviño et al., 2017). Balancing the protection of boreal forests and growing extraction of forest
69 resources for bioenergy and bio-products (following new bio-economy policy goals) requires
70 development of alternative ways to manage boreal forests (Hetemäki et al., 2017). Yet, the shift in
71 the order of priorities driving forest resources management is essential to obtain multifunctional
72 forest landscapes.

73 Mitigating the conflict between biodiversity conservation, the provision of non-timber ecosystem
74 services, and timber extraction requires application of less intensive forest management and/or
75 careful landscape planning (e.g., Eyvindson et al., 2018). Several alternative management techniques

76 that balance economic and ecological objectives have been recently developed. As such, they mimic
77 natural disturbances to emulate forest structures important for biodiversity (Kuuluvainen and
78 Grenfell, 2012) or reduce intensity of forest extraction spatially or temporally (Hanski, 2011). This
79 can be implemented by delaying clear-felling, limiting thinning, conducting selective harvest or
80 simply by leaving areas unmanaged (Äijälä et al., 2014). Forest planning could be applied through
81 spatial allocation of intensive and less intensive resource extraction, such as land-sharing and land
82 sparing approaches (Edwards et al., 2014; Messier et al., 2009). Considering the potential conflict
83 between resource extraction and habitat availability for threatened species, and the diversity of life
84 forms in forests, it is unlikely that a single forest management alternative systematically applied at
85 large scale would support multifunctional landscape (Haight and Monserud, 1990). Contrary, a
86 diverse range of management approaches may lead to a diverse forest structure and support forest
87 multifunctionality (Mönkkönen et al., 2014; Triviño et al., 2017). Yet, ecosystem services are
88 provided at various spatial scales, and the planning scale should match or be larger than the scale
89 services are provided (Pohjanmies et al., 2019; Raudsepp-Hearne and Peterson, 2016).

90 Specific forest management alternatives have been recommended for their ability to provide specific
91 ESS. In the last few decades, rotation forestry has been clearly the dominant method for timber
92 extraction throughout the boreal forest, as well as in large areas of planted forests in temperate
93 regions (Appelroth et al., 1948)(Appelroth et al., 1948). Since 1950s, intensive practices using clear-
94 cut harvesting resulted in impoverished stand structural diversity, fragmented forest structures, and
95 lowered structural variability at the landscape scale in most forests in Fennoscandia (Kuuluvainen et
96 al., 2012). Alternatively, continuous cover forestry, which maintains a forest canopy at all times, and
97 does not use clear-felling, has received considerable attention for application in boreal and
98 temperate forests (Pukkala and Gadow, 2012). In Fennoscandia, selective logging of individual large
99 trees that reached a certain size (target diameter harvesting) is among others the most applied
100 silvicultural system for continuous cover forestry. Recent research compared selective logging
101 (further referred as continuous cover forestry – CCF) with clear-felling approaches (result of
102 traditional rotation forestry – RF) in a wide range of forest conditions (Peura et al., 2018; Pukkala et
103 al., 2011). These studies highlighted the potential for CCF to perform better in terms of providing
104 ecosystem services, biodiversity, and general multifunctionality than RF.

105 To reconcile the negative effects of long-term clear-cutting and the potential benefits of CCF, many
106 researchers and NGOs are advocating in favour of the latter to replace the former. For instance, a
107 citizen initiative (VN/1699/2018¹) in Finland aims to promote biodiversity and nature protection,
108 through fully banning clear-cut activities in State-owned forests. However, this could lead to a
109 consistent application of CCF management approaches throughout a forested landscape, and may
110 thereby homogenize the landscape, i.e. lower diversity of forest structures. In addition, the land-use
111 intensity and negative environmental impacts could be higher with consistent application of CCF
112 than with consistent application of RF. For a given amount of timber extraction, CCF as compared to
113 RF may be less intensive in space but more intensive in time; hence increasing frequency of human-
114 induced disturbances. This may potentially have negative impacts to biodiversity and ecosystem
115 services other than timber.

116 Efficient resource use and conservation efforts require careful planning, where a combinations of
117 management alternatives and their share over the landscape can fulfill specific management
118 objectives. Here we explore the trade-offs between management alternatives through an
119 optimization approach, focusing on efficient uses of forest resources. Restricting the range of the
120 management alternatives could reduce the efficiency of the overall management objectives. We
121 hypothesise that exclusive and consistent use of a single type of forest management will likely
122 reduce the full potential efficiency of the forest landscape to simultaneously deliver ESS and
123 maintain BD. Nevertheless, restricting some management options could facilitate the
124 implementation of optimal planning in the real world by reducing possibilities to choose from for
125 forest owner. Our study aims to evaluate the independent performance of consistent use of CCF or
126 RF management alternatives, compared to combinations of all available management alternatives in
127 providing landscape-level BD and ESS. We examine the entire range of land-use intensity by varying
128 the desired net present income (NPI) of the landscape from no income, landscape level set-aside
129 (SA) management to the maximal NPI revenues. Further, we evaluate the performance of the
130 scenarios in terms of their multifunctionality at the landscape level. Our multifunctionality metrics
131 include both BD and non-timber ESS indicators.

¹ <https://www.kansalaisaloite.fi/fi/aloite/3184>

132 Materials and Methods

133 *Forest data and simulations under alternative management alternatives*

134 Our study area represents a typical Finnish production forest landscape (see Fig. 1), consisting of
135 forest stands located within a single watershed in Central Finland. We used the forest stand
136 information from the Finnish Forest Centre that is publicly available (www.metsään.fi). The
137 watershed was used as a natural boundary consisting of 1,475 relatively structurally homogenous
138 forest stands over 2,242 ha. The growth and management of the forest was simulated using the
139 open-source forest simulator SIMO (Rasinmäki et al., 2009) for 100 years, separated into 20 five-year
140 periods. For each stand, we simulated a maximum of 58 management alternatives. The exact
141 number of management alternatives applied depends on the specific initial conditions of each
142 individual forest stand. In total, 17 possible variations were available for RF management, 40
143 variations for CCF management, and one alternative where no management actions (set-aside) were
144 taken in the forest. Variations in RF management included changes to the timing of final felling,
145 optional thinning, and increased green tree retention (see further details in Eyvindson et al., 2018). A
146 basic form of CCF management follows the set of rules identified in Äijälä et al. (2014). To create a
147 maximum of CCF alternatives, we varied two rules defining timing of harvesting. First, we varied the
148 pre-defined site-specific basal area (m^2/ha) requirement ($16 \text{ m}^2/\text{ha}$ for less fertile sites to $22 \text{ m}^2/\text{ha}$
149 for fertile sites) prior to harvesting by -3, ± 0 , +3, +6. Additionally, we varied the timing of the first
150 harvest in 5 year increments up to a delay of 45 years. The cutting cycle were afterwards determined
151 within the simulation based on basal area requirements. A summary of the management
152 alternatives is presented in Appendix A.

153

154 *Ecosystem services and biodiversity indicators*

155 We calculated indicators for four BD and ESS components at the stand level, based on available
156 models and the simulated structural characteristics of each stand. The four components reflect
157 important aspects for Finnish nature and people: i) recreational ecosystem services and non-timber
158 production; ii) climate change mitigation, iii) suitable habitat for terrestrial vertebrate biodiversity,
159 and iv) suitable habitat for red-listed species dependent on deadwood.

160 Recreational ESS and non-timber production included bilberry, mushrooms and scenic beauty.

161 Bilberry (*Vaccinium myrtillus*) is one of the most common wild berries in Finland and has high

162 recreational and commercial value (Vaara et al., 2013). Bilberry yield (kg) was estimated using the
163 models of Miina et al. (2016) which predicts yield based on stand characteristics such as age, basal
164 area and dominant tree species. Mushrooms have also both recreational and commercial value in
165 Finland (Peura et al., 2016). Marketed mushrooms yield (kg) was estimated using the models of
166 Tahvanainen et al. (2016). While the mushroom models were developed for Eastern Finland in
167 Spruce dominating stands, the model cannot provide highly accurate estimations for mushroom
168 yield (the models have a predictive capacity of 23%). Yet, they provide an indication on the
169 suitability of the sites for mycorrhizal mushrooms. Scenic beauty (no unit) was calculated using the
170 index developed by Pukkala et al. (1988), which estimates people's average opinion about the
171 recreational value and beauty of forests based on slides and computer drawings of managed stands.
172 The age and size of trees increased the recreational and beauty value as well as a big share of pines
173 and birches.

174 Climate change mitigation considered the mass of carbon contained within timber (kg C), dead wood
175 (kg C), and soil (kg C) as a proxy for carbon stock. Timber was calculated as the total volume of
176 standing timber from the different tree species. Dead wood volume (m³) was measured as the total
177 amount of dead wood from the different dead wood types comprising different tree species and
178 decay stages. Deadwood decomposition was modeled through five decay stages using
179 decomposition models from Mäkinen et al. (2006). To estimate soil carbon, for mineral soils we used
180 the models from Liski and Westman (1997) to provide initial soil carbon values, and to model the
181 development of soil carbon we used the Yasso07 modelling framework (Liski et al., 2005; Tuomi et
182 al., 2011, 2009). Drained peatland soils were modeled using the carbon flux models proposed by
183 Ojanen et al. (2014). In this study we do not include the potential carbon storage through long-
184 lasting wood products, as the forest landscape is our system boundary.

185 Suitable habitat for vertebrate biodiversity included the habitat availability for six species: western
186 capercaillie (*Tetrao urogallus*), siberian flying squirrel (*Pteromys volans*), hazel grouse (*Bonasia*
187 *bonasa*), long-tailed tit (*Aegithalos caudatus*), lesser-spotted woodpecker (*Dendrocopos minor*), and
188 three-toed woodpecker (*Picoides tridactylus*). We selected these species to represent a wide range
189 of habitat types, and diverse social and economic values including game birds, umbrella, and
190 threatened species. The habitat suitability models were taken from Mönkkönen et al. (2014).

191 Finally, we explored the suitable habitat availability for 27 red-listed species dependent on dead
192 wood (fungi and arthropods). Dead wood is a critical resource in boreal forests (Stokland et al.,
193 2012); a good indicator of forest biodiversity (Gao et al., 2015; Lassauce et al., 2011), and the lack of

194 dead wood is the most important threat for species in Finnish forests (Tikkanen et al., 2006). The
195 habitat suitability models were taken from Tikkanen et al. (2007). A total of six ESS and 33 BD criteria
196 were integrated into a multifunctionality assessment.

197 *Forest multifunctionality*

198 We explored forest multifunctionality as a landscape metric rather than a stand-level characteristic.
199 Therefore, all indicators were first evaluated at stand level and then aggregated over the study area
200 to produce the total value over the landscape. We measured the ability of the forest landscape to
201 maintain high levels of all ESS and BD components (van der Plas et al., 2016). We defined
202 multifunctionality as the sum of the four normalized components (eq. 1, standardized by theoretical
203 maximum and minimal values derived from the pay-off table, Table 1), with equal priorities between
204 the components of multifunctionality. We aggregated indicators within components through two
205 measures: as the average value between all indicators (eq. 2a) and as the minimum value across all
206 indicators (eq. 2b). For climate change mitigation and non-timber ESS, components were estimated
207 as the average (of equal importance) of their indicators (eq. 2a) while BD components were
208 estimated as the minimum value across the biodiversity indicators (eq. 2b). We rationale that : i) in
209 climate mitigation, carbon sequestration in deadwood can substitute carbon in standing timber; ii) in
210 non-timber ecosystem services, we maximize the summed production of these social benefits and iii)
211 for biodiversity, we want to preserve all species, hence maximize the habitat availability for the
212 species with lowest score. All species have an existence value, and we cannot thus assume that the
213 suffering of a single species can be offset by the success of other species.

214 To account for the increased costs of selective harvesting by the CCF alternatives, timber prices
215 obtained from CCF management are set to be 75% of estimated price of RF. This adjustment reflects
216 a doubling in harvesting costs per m³, while CCF management extract approximately 50% of
217 harvested timber than RF operations. As discount rate for the NPI, we considered a factor of 2%,
218 which is often applied to cover long-term economic problems in forestry, and to reflect on increasing
219 discount rate we examined a 4% rate in Appendix B. The NPI was chosen as economic indicator as it
220 does not account for the remaining standing timber values under set aside, where forest values are
221 rather important for conservation reasons.

222 Through the computational material, readers can explore the use of average or minimum value used
223 in combination for all components (gitlab.jyu.fi/kyjoeyvi/multifunctionality_costs). The
224 mathematical translation of these choices is shown in more detail in the following section.

225 *Formulation of the optimization problem*

226 Through an optimization framework we explore the trade-offs between the net present income
 227 (NPI) obtained through harvesting operations and forest multifunctionality. We have opted to use
 228 NPI as the economic value of the forest, as this is how Metsähallitus (the Finnish governmental
 229 organization managing state owned forests) selects stands to harvest. The higher NPI values
 230 represent higher intensity of timber extraction. The optimization process was performed three
 231 times: i) including all management alternatives, ii) including only RF management alternatives, and
 232 iii) including only CCF management alternatives.

233 The general frame for the optimization problem is one where we maximize multifunctionality (eq. 1),
 234 subject to a constraint where NPI meets or exceeds a particular targeted value (eq. 5). This
 235 optimization can be seen as a goal programming formulation (such as in Eyvindson, 2012), where
 236 different components can be treated with different distance measures. The proposed objective
 237 function is:

$$[1] \quad \max \sum_{b \in B} \frac{(D_b - D_{b^*})}{(D_b^* - D_{b^*})}$$

238 subject to:

$$[2a] \quad D_b = \frac{1}{\#T_b} \sum_{t \in T_b} \frac{(f_t - f_{t^*})}{(f_t^* - f_{t^*})}$$

$$[2b] \quad D_b = \operatorname{argmin}_{t \in T_b} \frac{(f_t - f_{t^*})}{(f_t^* - f_{t^*})}, \forall b \in B$$

$$[3] \quad f_t = \sum_{p \in P} \left(\frac{\sum_{j \in J} \sum_{k=1}^{K_j} x_{jk} Z_{jkp}^t}{\#P} \right)$$

$$[4] \quad f_{NPV} = \sum_{p \in P} \left(\frac{\sum_{j \in J} \sum_{k=1}^{K_j} x_{jk} Z_{jkp}^I}{(1+r)^{(2.5+(p-1)*5)}} \right)$$

$$[5] \quad f_{NPI} \geq q * f_{NPI}^*$$

$$[6] \quad \sum_{k=1}^{K_j} x_{jk} = 1, \forall j = 1, \dots, J$$

$$[7] \quad q \in (0,1), x_{jk} \in [0,1], \forall j \in J, k \in K$$

239 where D_b, D_b^* and D_{b*} represents the measured, *ideal and anti-ideal* deviation for component b ; B is
 240 the set of components, f_t^*, f_{t*} and f_t respectively represent the ideal, anti-ideal and obtained value
 241 for indicator t ; f_{NPI} is the value for NPI; T_b is the set of indicators in component b , x_{jk} is the decision
 242 to harvest stand j according to management alternative k ; K_j is the set of management types for
 243 stand j ; z_{jkp}^t is the value of indicator t associated with conducting management alternative k on
 244 stand j during period p ; P is the set of periods under consideration; r is a parameter for the discount
 245 rate, and q is a parameter that determines the required proportion of the maximum net present
 246 income. To calculate the ideal and anti-ideal values, a series of separate optimization problem was
 247 run both maximizing and minimizing the single indicator using all feasible management alternatives.

248 Multifunctionality is measured at the landscape level indicating the sum of specific normalized
 249 distances for each component. To normalize each component, we calculated a payoff table by
 250 independently optimizing the components, with and without the NPI constraint. This identifies the
 251 trade-offs between component groups and the range each multifunctionality measure can take. The
 252 ideal and anti-ideal values (D_b^* and D_{b*}) were extracted from that payoff table (Table 1). We
 253 assessed multifunctionality as aggregate of the distance values from each of the four components.
 254 Distance was measured in two ways, using the L^1 distance (also known as the Manhattan distance)
 255 and the as the L^∞ distance (also known as Chebyshev distance). These measures have a preferential
 256 translation, where L^1 distance measures the efficiency amongst criteria while L^∞ measures equity
 257 between criteria (Diaz-Balteiro et al., 2013).

258 For this problem formulation, the objective (Eq. [1]) maximizes the summed normalized distance
 259 from each component of multifunctionality. Eq. [2] measures the distance of each component of the
 260 multifunctionality, where 2a measures the distance for non-timber ESS and carbon storage using the
 261 L^1 distance metric while 2b measures the L^∞ distance for BD. Each component of multifunctionality is
 262 measured by either of these equations, depending on how the components of multifunctionality are
 263 measured. Eq. [3] evaluates the obtained landscape level value for the specific criterion t . Eq. [4]
 264 evaluates the obtained NPI for the landscape. Eq. [5] establishes a required minimum obtained NPI.
 265 Eq. [6] is the constraint requiring that each stand has some form of management alternative used.
 266 Eq. [7] sets the range of values for the parameters and decision variables. All variables used in this
 267 problem formulation are described in Table 2. The optimisation problem was solved using Pyomo
 268 (Hart et al., 2011) in conjunction with both CPLEX and CBC (Forrest et al., 2018). To allow for

269 replication we uploaded the code on an online repository together with a sample dataset
270 (gitlab.jyu.fi/kyjoeyvi/multifunctionality_costs).

271 Results

272 For each scenario, the proportion of unmanaged forested areas decrease following a negative linear
273 trend as the monetary value extraction increases (Fig. 2). Irrespective of the land-use intensity
274 (represented as increasing timber extraction, and measured as NPI), CCF scenario always
275 outperforms RF scenario in terms of overall landscape multifunctionality (Fig. 3a). CCF scenarios
276 provide corresponding multifunctionality values to the scenario where all management options are
277 allowed, at low and intermediate land-use intensities ($NPI < 5 \text{ k€ / ha}$). Only at high timber extraction
278 levels, excluding the RF from forest management alternatives caused multifunctionality losses (CCF
279 relative to all management types). At maximal NPI, a consistent use of CCF results in about half of
280 the multifunctionality reduction than relying consistently on the RF alternatives. In other words, if all
281 management options are allowed, CCF is a prevailing forest management method except at high
282 levels of land-use intensity, where it is optimal to combine CCF and RF when targeting
283 multifunctionality (Fig. 2).

284 If solely RF management alternatives are applied, multifunctionality monotonically decline with
285 increasing land-use intensity (Fig. 3a). This trend in overall multifunctionality stems from the
286 continuous decrease of non-timber ESS, carbon storage, and vertebrate BD components. (Fig. 3b-d.).
287 Deadwood BD exhibited a dampened humped shape curve, peaking at about 6k €/ha (Fig. 3e.).

288 Under CCF and all management alternative scenarios, the pattern for overall multifunctionality is
289 unimodal, and maximum multifunctionality values are achieved with an intermediate attainment of
290 NPI (approximately 4 k €/ha, Fig. 3a). The pattern is likely because of the BD components of
291 multifunctionality, while the provision of non-timber ESSs remains relatively stable and the carbon
292 storage declines steadily with increasing NPI (Fig. 3b-e.).

293 Individual non-timber ESS and vertebrate habitat suitability indicators show contrasting patterns
294 along the timber extraction intensity gradient irrespective of whether RF or CCF management is
295 applied. This suggests conflict among the indicators, and shows that there is much variation in terms
296 whether CCF is better than RF, or vice versa. This trend is also seen in the payoff table (Table 1), as
297 the range and variation between the components is similar to the trade-off seen in the scenario
298 analysis. The development of the dead wood dependent species is interesting, as the set of 27
299 indicator species seem to follow one of two trends (Fig. 3e). These species seemingly either prefer

300 forests that receive no forest management or they prefer moderate management actions. This trend
301 is very similar between only using CCF or RF. However, maximum value is reached with CCF and the
302 optimum for RF is at higher NPI than for CCF.

303 Discussion

304 The results of this study highlight the significant potential for conflicts between timber extraction
305 and forest multifunctionality. Within selected indicators, we found negative effects of timber
306 extraction on deadwood habitat indicators, scenic beauty, and carbon storage. On the other hand,
307 harvesting can positively affect a small subset of the indicators such as mushrooms yield, and both
308 continuous cover forestry (CCF) and rotation forestry (RF) showed initial positive trend for some
309 dead wood habitat indicators. The complexity of how individual species groups respond to
310 extraction levels and forest management alternatives increases with an increasing number of species
311 considered. Some vertebrate species benefits from CCF, other vertebrate species can be maintained
312 using RF until the requirement for NPI exceeds a specific level. Yet, the siberian flying squirrel's
313 habitat decreased with increasing timber extraction level regardless of the applied harvesting system
314 as this endangered species inhabits old spruce-dominated mixed forests (Wistbacka et al., 2018).

315 As a political tool for improving conservation practices, restricting forest management alternatives
316 may not be a fully justifiable position. In this case, if we restrict the range of usable forest
317 management alternatives to either CCF or RF, both economic and ecological outcomes may either
318 remain similar or perform more poorly than if managers have all options available. However, this
319 analysis is based on the use of optimization, and implies that managers are making well-informed
320 decisions regarding both the economic and ecological performance of the forest, and that all forest
321 owners have a consistent preference for non-timber ecosystem services and biodiversity protection.
322 Forest managers may utilize heuristic optimization (Gigerenzer and Gaissmaier, 2011), or follow
323 simple rules to strategize forest management planning (Äijälä et al., 2014). Unless the forest
324 management planning relies on up to date scientific evidence, the overall timber and overall forest
325 functioning will likely be suboptimal.

326 The Finnish case study highlights the positive impact from the recent legislative change lifting the
327 ban of practicing CCF. Until a recent legislative change in the Finnish forest act (2014)², forest
328 owners had been restricted to intensively manage their forests and extract their timber using a form
329 of clear-felling (Appelroth et al., 1948). However, psychological barriers may prevent forest owners
330 from applying CCF due to a lack of familiarity, preventing the most appropriate management option
331 to be selected for a specific forested area (Isoaho et al., 2019). Yet, CCF methods are still not widely
332 applied. The recent citizen initiative strives to restrict the use of RF in Finnish State-owned forests
333 (~9.1 M ha of which ~85% are located in Northern Finland), while respects private forest owner's
334 decision-making capabilities. The RF restriction initiative aimed to support conservation efforts. If
335 high revenue targets are required from Metsähallitus (the Finnish governmental organization
336 managing state owned forests), exclusive reliance on CCF will have a slight positive impact on
337 ecosystem services and biodiversity considerations, as spatially intensive harvesting would be
338 replaced by temporally intensive harvesting.

339 The analysis we present highlights the potential benefits of utilizing a diverse range of management
340 alternatives compared to single applied management (Haight and Monserud, 1990). The use of CCF
341 plays an important role in enhancing BD and ESS features while contributing significant economic
342 value (Díaz-Yáñez et al., 2020; Pukkala, 2016). However, our modelling approach contains substantial
343 uncertainties which may have a dramatic impact on the provisions of BD and ESS, and the possible
344 economic output from the forests. As CCF has been used on very limited areas in Fennoscandia, and
345 for a limited amount of time, scientific knowledge on landscape-scale CCF management is lacking. As
346 compared to RF, the modelling of growth, natural regeneration, and mortality under CCF might have
347 larger errors, as large scale, systematic sampling of this management approach has not yet been
348 performed. The economical profitability of CCF or RF depends on the initial conditions of the forest
349 stands and the respective costs of wood procurement. CCF is usually more profitable for less
350 productive stands and can be more profitable even with a sizeable increase in wood procurement
351 costs (Rämö and Tahvonen, 2017) (in our study ~13€ per m³ for log wood and ~7 € per m³ for
352 pulpwood).

² <https://www.finlex.fi/fi/laki/kaannokset/1996/en19961093.pdf>

353 There are several reasons why CCF can be more profitable than RF: i) Log/pulp ratio: CCF provides
354 more log and less pulp wood than RF, as the thinning is done from above, extracting the biggest
355 trees, instead than from below, extracting the smallest trees like in RF; ii) Regeneration method:
356 CCF assumes that there is a natural regeneration whereas in RF the regeneration is artificial by
357 planting new trees which is has a high economic cost. It is uncertain, however, if the natural
358 regeneration is always successful in CCF; iii) Discount rate: this has an influence on the timing of
359 timber harvests and expected rotation lengths of forests (Brukas et al., 2001). Changes in discount
360 rates may change the share of the landscape managed under RF and CCF management alternatives
361 (see Appendix B), where high discount rates reduce the supply of non-timber ESS compared to low
362 discount rates (Pukkala, 2016).

363 The use of forest planning methods and optimization can provide an optimistic view on how
364 harvesting actions can balance between timber extraction and landscape-level multifunctionality.
365 However, our approach relies on a single climate alternative and neglects potential disturbances
366 throughout the 100-year time horizon. In boreal forests, continuing water availability and increasing
367 temperatures under climate change will likely increase forest growth rates (Kellomäki, 2017). In
368 addition, climate change might increase the risk of wind damage through the shortening of the
369 periods of frozen soil and releasing tree root anchorage during the windiest time of the year (Peltola
370 et al., 2010). Warmer winters may also increase risks of insect outbreaks (Neuvonen and Viiri, 2017),
371 or potential development of newcomer forest pest species, such as *Ips amitius* (Økland et al., 2019).
372 Therefore, omitting disturbances from the forest management planning might overestimate
373 expected revenues (Díaz-Yáñez et al., 2019). We acknowledge that the impacts of climate change
374 and disturbance will affect our results. However, we believe that the consequences of climate
375 change on tree growth and disturbance risk will be equally distributed between CCF and RF
376 management alternatives. Additionally, we anticipate the increased disturbances may have a
377 stronger impact on RF than on CCF management alternatives. This will likely be due to several
378 factors. CCF is likely to have less canopy height variation between stands, i.e., avoiding open edge
379 stands protecting against wind (Zubizarreta-Gerendiain et al., 2019, 2016), and the stands will likely
380 have a higher mixing of species, mitigating potential pest outbreaks (Hlásny et al., 2019). Wider
381 range of applied management regimes increases landscape multifunctionality and compositional
382 diversity. This might provide a buffer against uncertainties and possible disturbances, compared to
383 single objective, or highly correlated ESS management types (Knoke et al., 2016).

384 Conclusion

385 From a forest planning perspective, limiting the diversity of management options will limit the ability
386 of the forest to attain a full potential of multifunctional benefits, especially at high extraction level.
387 Restricting management (either restricting RF or CCF) will likely lower the economic value, and
388 landscape multifunctionality. Thus, achieving an efficient solution between multifunctionality and
389 economic benefit will require a diverse set of management alternatives, utilizing primarily CCF with
390 small share of RF management. Interestingly, in Fennoscandian forest landscapes under natural
391 disturbance regimes, the proportion of stand replacing disturbances has been between 20 – 30%,
392 and cohort dynamics (in pine dominated forests) or gap dynamics (in spruce dominated forests) have
393 been dominating (Kuuluvainen and Aakala, 2011). Thus, from the point of view of mimicking natural
394 disturbance dynamics, rotation forestry, which emulates structures typical for stands after stand-
395 replacing disturbances, should be secondary to continuous cover forestry, which in turn better
396 emulate fine-scale disturbances. According to our results, to maximize multifunctionality while
397 obtaining high timber extraction rates, the utilization of RF should be between 10 – 25% of the total
398 forest area. In the boreal forests, the primarily forest management alternative applied is RF,
399 reductions in clear cuts would likely improve landscape-scale forest multifunctionality, including
400 non-timber ecosystem services and biodiversity. However, as large proportion of productive forests
401 in Fennoscandia are privately owned, encouraging CCF in these would also be required to improve
402 landscape multifunctionality. On the other hand, complete restriction of RF in State-owned forests,
403 as suggested in Finland, will likely impede the development of the full potential of multifunctional
404 landscape, particularly in the era of bioeconomy and its expected high timber demands.

405 Acknowledgements:

406 This work was supported by the Finnish Ministry of Agriculture and Forestry and by the European
407 Union through the programs ERA-NET SumForest program (project: FutureBioEcon — Sustainable
408 future of European Forests for developing the bioeconomy, Forest Values (project MultiForest -
409 Management for multifunctionality in European forests in the era of bioeconomy), and Biodiversa
410 (project: BioESSHealth - Scenarios for biodiversity and ecosystem services acknowledging health),
411 and through the the Academy of Finland (project number 275329). We thank members of the BERG
412 team (www.jyu.fi/berg) for their comments on the manuscript. RD and MT were supported by
413 postdoctoral fellowships from the Kone Foundation.

414 **Author contributions:**

415 **Kyle Eyvindson:** Conceptualization, Methodology, software, Writing – Original draft, **Rémi Duflot:**
416 Investigation, Visualization, Writing – Review & Editing, **María Triviño:** Conceptualization, Writing –

417 Review & Editing, Visualization, **Clemens Blattert**: Writing – Review & Editing, Formal Analysis,
418 **Mária Potterf**: Writing – Review & Editing, Investigation, **Mikko Mönkkönen**: Conceptualization,
419 Writing – Review & Editing, Funding acquisition

420 References:

- 421 Äijälä, O., Koistinen, A., Sved, J., Vanhatalo, K., Väisänen, P., 2014. Hyvän metsänhoidon
422 suositukset [Good forest management recommendations]. Forestry Development Center Tapio
423 [In Finnish].
- 424 Appelroth, E., Heikinheimo, O., Kalela, E., Laitakari, E., Lindfors, J., Sarvas, R., 1948. Julkilausuma.
425 Metsätaloudellinen Aikakausl. 65, 315–316.
- 426 Brukas, V., Jellesmark Thorsen, B., Helles, F., Tarp, P., 2001. Discount rate and harvest policy:
427 implications for Baltic forestry. *For. Policy Econ.* 2, 143–156. [https://doi.org/10.1016/S1389-](https://doi.org/10.1016/S1389-9341(01)00050-8)
428 [9341\(01\)00050-8](https://doi.org/10.1016/S1389-9341(01)00050-8)
- 429 Díaz-Balteiro, L., González-Pachón, J., Romero, C., 2013. Goal programming in forest management:
430 customising models for the decision-maker's preferences. *Scand. J. For. Res.* 28, 166–173.
431 <https://doi.org/10.1080/02827581.2012.712154>
- 432 Díaz-Yáñez, O., Arias-Rodil, M., Mola-Yudego, B., González-Olabarria, J.R., Pukkala, T., 2019.
433 Simulating the effects of wind and snow damage on the optimal management of Norwegian
434 spruce forests. *For. An Int. J. For. Res.* 92, 406–416. <https://doi.org/10.1093/forestry/cpz031>
- 435 Díaz-Yáñez, O., Pukkala, T., Packalen, P., Peltola, H., 2020. Multifunctional comparison of different
436 management strategies in boreal forests. *For. An Int. J. For. Res.* 93, 84–95.
437 <https://doi.org/10.1093/forestry/cpz053>
- 438 Díaz, S., Settele, J., Brondízio, E., Ngo, H., Guèze, M., Agard, J., Arneith, A., Balvanera, P., Brauman,
439 K., Butchart, S., Chan, K., Garibaldi, L., Ichii, K., Liu, J., Subrmanian, S., Midgley, G.,
440 Miloslavich, P., Molnár, Z., Obura, D., Pfaff, A., Polasky, S., Purvis, A., Razzaque, J., Reyers,
441 B., Chowdhury, R., Shin, Y., Visseren-Hamakers, I., Willis, K., Zayas, C., 2019. Summary for
442 policymakers of the global assessment report on biodiversity and ecosystem services of the
443 Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.
- 444 Edwards, D.P., Gilroy, J.J., Woodcock, P., Edwards, F.A., Larsen, T.H., Andrews, D.J.R., Derhé,
445 M.A., Docherty, T.D.S., Hsu, W.W., Mitchell, S.L., Ota, T., Williams, L.J., Laurance, W.F.,
446 Hamer, K.C., Wilcove, D.S., 2014. Land-sharing versus land-sparing logging: reconciling timber
447 extraction with biodiversity conservation. *Glob. Chang. Biol.* 20, 183–191.
448 <https://doi.org/10.1111/gcb.12353>
- 449 Eyvindson, K., 2012. Balancing equity and efficiency of goal programming for use in forest
450 management planning. *Can. J. For. Res.* 42, 1919–1925. <https://doi.org/10.1139/x2012-135>
- 451 Eyvindson, K., Repo, A., Mönkkönen, M., 2018. Mitigating forest biodiversity and ecosystem service
452 losses in the era of bio-based economy. *For. Policy Econ.* 92, 119–127.
453 <https://doi.org/10.1016/J.FORPOL.2018.04.009>
- 454 Forrest, J., Ralphs, T., Vigerske, S., Hafer, L., Kristjansson, B., Jpfasano, Straver, E., Lubin, M.,
455 Santos, H.G., Rlougee, Saltzmann, M., 2018. coin-or/Cbc [WWW Document].
456 <https://doi.org/10.5281/zenodo.1317566>
- 457 Gao, T., Nielsen, A.B., Hedblom, M., 2015. Reviewing the strength of evidence of biodiversity
458 indicators for forest ecosystems in Europe. *Ecol. Indic.* 57, 420–434.
459 <https://doi.org/http://dx.doi.org/10.1016/j.ecolind.2015.05.028>
- 460 Gigerenzer, G., Gaissmaier, W., 2011. Heuristic Decision Making. *Annu. Rev. Psychol.* 62, 451–482.
461 <https://doi.org/10.1146/annurev-psych-120709-145346>

- 462 Haight, R.G., Monserud, R.A., 1990. Optimizing any-aged management of mixed-species stands: II.
463 Effects of decision criteria, *Forest Science* 36(1):125-144.
- 464 Hansen, M.C., Stehman, S. V, Potapov, P. V, 2010. Quantification of global gross forest cover loss.
465 *Proc. Natl. Acad. Sci.* 107, 8650–8655. <https://doi.org/10.1073/pnas.0912668107>
- 466 Hanski, I., 2011. Habitat loss, the dynamics of biodiversity, and a perspective on conservation. *Ambio*
467 40, 248–55.
- 468 Harrison, P.A., Vandewalle, M., Sykes, M.T., Berry, P.M., Bugter, R., de Bello, F., Feld, C.K., Grandin,
469 U., Harrington, R., Haslett, J.R., Jongman, R.H.G., Luck, G.W., da Silva, P.M., Moora, M.,
470 Settele, J., Sousa, J.P., Zobel, M., 2010. Identifying and prioritising services in European
471 terrestrial and freshwater ecosystems. *Biodivers. Conserv.* 19, 2791–2821.
472 <https://doi.org/10.1007/s10531-010-9789-x>
- 473 Hart, W.E., Watson, J.P., Woodruff, D.L., 2011. Pyomo: Modeling and solving mathematical programs
474 in Python. *Math. Program. Comput.* 3, 219–260. <https://doi.org/10.1007/s12532-011-0026-8>
- 475 Hetemäki, L., Hanewinkel, M., Muys, B., Ollikainen, M., Palahí, M., Trasobares, A., 2017. Leading the
476 way to a European circular bioeconomy strategy. *From Science to Policy* 5.
- 477 Hlásny, T., Krokene, P., Liebhold, A., Montagné-Huck, C., Müller, J., Qin, H., Raffa, K., Schelhaas,
478 M.-J., Seidl, R., Svoboda, M., 2019. Living with bark beetles: impacts, outlook and management
479 options. p. From Science to Policy 8. European Forest Institut.
- 480 Isoaho, K., Burgas, D., Janasik, N., Mönkkönen, M., Peura, M., Hukkinen, J.I., 2019. Changing forest
481 stakeholders' perception of ecosystem services with linguistic nudging. *Ecosyst. Serv.* 40,
482 101028. <https://doi.org/10.1016/J.ECOSER.2019.101028>
- 483 Kellomäki, S., 2017. *Managing Boreal Forests in the Context of Climate Change: Impacts, Adaptation*
484 *and Climate Change Mitigation.* CRC Press.
- 485 Knoke, T., Paul, C., Hildebrandt, P., Calvas, B., Castro, L.M., Hartl, F., Dollerer, M., Hamer, U.,
486 Windhorst, D., Wiersma, Y.F., Curatola Fernández, G.F., Obermeier, W.A., Adams, J., Breuer,
487 L., Mosandl, R., Beck, E., Weber, M., Stimm, B., Haber, W., Fürst, C., Bendix, J., 2016.
488 Compositional diversity of rehabilitated tropical lands supports multiple ecosystem services and
489 buffers uncertainties. *Nat. Commun.* 7, 1–12. <https://doi.org/10.1038/ncomms11877>
- 490 Kuuluvainen, T., Aakala, T., 2011. Natural forest dynamics in boreal Fennoscandia: a review and
491 classification. *Silva Fenn.* 45, 823–841.
- 492 Kuuluvainen, T., Grenfell, R., 2012. Natural disturbance emulation in boreal forest ecosystem
493 management — theories, strategies, and a comparison with conventional even-aged
494 management. *Can. J. For. Res.* 42, 1185–1203. <https://doi.org/10.1139/X2012-064>
- 495 Kuuluvainen, T., Tahvonen, O., Aakala, T., 2012. Even-aged and uneven-aged forest management in
496 boreal Fennoscandia: a review. *Ambio* 41, 720–37. <https://doi.org/10.1007/s13280-012-0289-y>
- 497 Lassauce, A., Paillet, Y., Jactel, H., Bouget, C., 2011. Deadwood as a surrogate for forest
498 biodiversity: Meta-analysis of correlations between deadwood volume and species richness of
499 saproxylic organisms. *Ecol. Indic.* 11, 1027–1039.
500 <https://doi.org/http://dx.doi.org/10.1016/j.ecolind.2011.02.004>
- 501 Liski, J., Palosuo, T., Peltoniemi, M., Sievänen, R., 2005. Carbon and decomposition model Yasso for
502 forest soils. *Ecol. Modell.* 189, 168–182. <https://doi.org/10.1016/J.ECOLMODEL.2005.03.005>
- 503 Liski, J., Westman, C.J., 1997. Carbon storage in forest soil of Finland. 2. Size and regional pattern.
504 *Biogeochemistry* 36, 261–274. <https://doi.org/10.1023/A:1005742523056>
- 505 Mäkinen, H., Hynynen, J., Siitonen, J., Sievänen, R., 2006. Predicting the decomposition of Scots
506 pine, Norway spruce, and birch stems in Finland. *Ecol. Appl.* 16, 1865–1879.
507 <https://doi.org/10.2307/40061757>

- 508 Messier, C., Bauhus, J., Doyon, F., Maure, F., Sousa-Silva, R., Nolet, P., Mina, M., Aquilué, N., Fortin,
509 M.-J., Puettmann, K., 2019. The functional complex network approach to foster forest resilience
510 to global changes. *For. Ecosyst.* 6, 21. <https://doi.org/10.1186/s40663-019-0166-2>
- 511 Messier, C., Tittler, R., Kneeshaw, D.D., Gélinas, N., Paquette, A., Berninger, K., Rheault, H., Meek,
512 P., Beaulieu, N., 2009. TRIAD zoning in Quebec: Experiences and results after 5 years. *For.*
513 *Chron.* 85, 885–896. <https://doi.org/10.5558/tfc85885-6>
- 514 Miina, J., Pukkala, T., Kurttila, M., 2016. Optimal multi-product management of stands producing
515 timber and wild berries. *Eur. J. For. Res.* 135, 781–794. <https://doi.org/10.1007/s10342-016-0972-9>
- 517 Mönkkönen, M., Burgas, D., Eyvindson, K., Le Tortorec, E., Peura, M., Pohjanmies, T., Repo, A.,
518 Triviño, M., 2018. Solving conflicts among conservation, economic, and social objectives in
519 boreal production forest landscapes: Fennoscandian perspectives, in: Perera, A. (Ed.),
520 *Ecosystem Services from Forest Landscapes: Broad-scale Considerations*. Springer, pp. 169–
521 219. https://doi.org/10.1007/978-3-319-74515-2_7
- 522 Mönkkönen, M., Juutinen, A., Mazziotta, A., Miettinen, K., Podkopaev, D., Reunanen, P., Salminen,
523 H., Tikkanen, O.-P., 2014. Spatially dynamic forest management to sustain biodiversity and
524 economic returns. *J. Environ. Manage.* 134, 80–89.
525 <https://doi.org/http://dx.doi.org/10.1016/j.jenvman.2013.12.021>
- 526 Neuvonen, S., Viiri, H., 2017. Changing climate and outbreaks of forest pest insects in a cold northern
527 country, Finland, in: Latola, K., Savela, H. (Eds.), *The Inter-Connected Arctic*. Springer, Cham,
528 pp. 49–59. https://doi.org/10.1007/978-3-319-57532-2_5
- 529 Newbold, T., Hudson, L.N., Hill, S.L.L., Contu, S., Lysenko, I., Senior, R.A., Börger, L., Bennett, D.J.,
530 Choimes, A., Collen, B., Day, J., De Palma, A., Díaz, S., Echeverria-Londoño, S., Edgar, M.J.,
531 Feldman, A., Garon, M., Harrison, M.L.K., Alhusseini, T., Ingram, D.J., Itescu, Y., Kattge, J.,
532 Kemp, V., Kirkpatrick, L., Kleyer, M., Correia, D.L.P., Martin, C.D., Meiri, S., Novosolov, M., Pan,
533 Y., Phillips, H.R.P., Purves, D.W., Robinson, A., Simpson, J., Tuck, S.L., Weiher, E., White, H.J.,
534 Ewers, R.M., Mace, G.M., Scharlemann, J.P.W., Purvis, A., 2015. Global effects of land use on
535 local terrestrial biodiversity. *Nature* 520, 45–50. <https://doi.org/10.1038/nature14324>
- 536 Ojanen, P., Lehtonen, A., Heikkinen, J., Penttilä, T., Minkkinen, K., 2014. Soil CO₂ balance and its
537 uncertainty in forestry-drained peatlands in Finland. *For. Ecol. Manage.* 325, 60–73.
538 <https://doi.org/10.1016/j.foreco.2014.03.049>
- 539 Økland, B., Flø, D., Schroeder, M., Zach, P., Cocos, D., Martikainen, P., Siitonen, J., Mandelshtam,
540 M.Y., Musolin, D.L., Neuvonen, S., Vakula, J., Nikolov, C., Lindelöw, Å., Voolma, K., 2019.
541 Range expansion of the small spruce bark beetle *Ips amitinus*: a newcomer in northern Europe.
542 *Agric. For. Entomol.* 21, 286–298. <https://doi.org/10.1111/afe.12331>
- 543 Peltola, H., Ikonen, V.-P., Gregow, H., Strandman, H., Kilpeläinen, A., Venäläinen, A., Kellomäki, S.,
544 2010. Impacts of climate change on timber production and regional risks of wind-induced
545 damage to forests in Finland. *For. Ecol. Manage.* 260, 833–845.
546 <https://doi.org/10.1016/J.FORECO.2010.06.001>
- 547 Peura, M., Burgas, D., Eyvindson, K., Repo, A., Mönkkönen, M., 2018. Continuous cover forestry is a
548 cost-efficient tool to increase multifunctionality of boreal production forests in Fennoscandia.
549 *Biol. Conserv.* 217, 104–112. <https://doi.org/10.1016/J.BIOCON.2017.10.018>
- 550 Peura, M., Triviño, M., Mazziotta, A., Podkopaev, D., Juutinen, A., Mönkkönen, M., 2016. Managing
551 boreal forests for the simultaneous production of collectable goods and timber revenues. *Silva*
552 *Fenn.* 50. <https://doi.org/10.14214/sf.1672>
- 553 Pimm, S.L., Jenkins, C.N., Abell, R., Brooks, T.M., Gittleman, J.L., Joppa, L.N., Raven, P.H., Roberts,
554 C.M., Sexton, J.O., 2014. The biodiversity of species and their rates of extinction, distribution,
555 and protection. *Science* 344, 1246752. <https://doi.org/10.1126/science.1246752>
- 556 Pohjanmies, T., Eyvindson, K., Mönkkönen, M., 2019. Forest management optimization across spatial

- 557 scales to reconcile economic and conservation objectives. *PLoS One* 14, e0218213.
558 <https://doi.org/10.1371/journal.pone.0218213>
- 559 Pohjanmies, T., Triviño, M., Le Tortorec, E., Salminen, H., Mönkkönen, M., 2017. Conflicting
560 objectives in production forests pose a challenge for forest management. *Ecosyst. Serv.* 28,
561 298–310. <https://doi.org/10.1016/J.ECOSER.2017.06.018>
- 562 Pukkala, T., 2016. Which type of forest management provides most ecosystem services? *For.*
563 *Ecosyst.* 3, 9. <https://doi.org/10.1186/s40663-016-0068-5>
- 564 Pukkala, T., Gadow, K. V., 2012. Continuous Cover Forestry. Book Series Managing Forest
565 Ecosystems, 24.
- 566 Pukkala, T., Kellomäki, S., Mustonen, E., 1988. Prediction of the amenity of a tree stand. *Scand. J.*
567 *For. Res.* 3, 533–544. <https://doi.org/10.1080/02827588809382538>
- 568 Pukkala, T., Lähde, E., Laiho, O., Salo, K., Hotanen, J.-P., 2011. A multifunctional comparison of
569 even-aged and uneven-aged forest management in a boreal region. *Can. J. For. Res.* 41, 851–
570 862. <https://doi.org/10.1139/x11-009>
- 571 Rämö, J., Tahvonen, O., 2017. Optimizing the harvest timing in continuous cover forestry. *Environ.*
572 *Resour. Econ.* 67, 853–868. <https://doi.org/10.1007/s10640-016-0008-4>
- 573 Rasinmäki, J., Mäkinen, A., Kalliovirta, J., 2009. SIMO: An adaptable simulation framework for
574 multiscale forest resource data. *Comput. Electron. Agric.* 66, 76–84.
575 <https://doi.org/10.1016/j.compag.2008.12.007>
- 576 Raudsepp-Hearne, C., Peterson, G.D., 2016. Scale and ecosystem services: how do observation,
577 management, and analysis shift with scale—lessons from Québec. *Ecol. Soc.* 21, art16.
578 <https://doi.org/10.5751/ES-08605-210316>
- 579 Rist, L., Felton, A., Nyström, M., Troell, M., Sponseller, R.A., Bengtsson, J., Österblom, H., Lindborg,
580 R., Tidåker, P., Angeler, D.G., Milestad, R., Moen, J., 2014. Applying resilience thinking to
581 production ecosystems. *Ecosphere* 5, art73. <https://doi.org/10.1890/ES13-00330.1>
- 582 Schwenk, W.S., Donovan, T.M., Keeton, W.S., Nunery, J.S., 2012. Carbon storage, timber
583 production, and biodiversity: comparing ecosystem services with multi-criteria decision analysis.
584 *Ecol. Appl.* 22, 1612–1627. <https://doi.org/10.1890/11-0864.1>
- 585 Stokland, J.N., Siitonen, J., Jonsson, B.G., 2012. Biodiversity in Dead Wood. Cambridge University
586 Press, Cambridge, UK.
- 587 Tahvanainen, V., Miina, J., Kurttila, M., Salo, K., 2016. Modelling the yields of marketed mushrooms
588 in *Picea abies* stands in eastern Finland. *For. Ecol. Manage.* 362, 79–88.
589 <https://doi.org/10.1016/j.foreco.2015.11.040>
- 590 Tikkanen, O.-P., Heinonen, T., Kouki, J., Matero, J., 2007. Habitat suitability models of saproxylic red-
591 listed boreal forest species in long-term matrix management: Cost-effective measures for multi-
592 species conservation. *Biol. Conserv.* 140, 359–372. <https://doi.org/10.1016/j.biocon.2007.08.020>
- 593 Tikkanen, O.-P., Martikainen, P., Hyvärinen, E., Junninen, K., Kouki, J., 2006. Red-listed boreal forest
594 species of Finland: associations with forest structure, tree species, and decaying wood. *Ann.*
595 *Zool. Fennici* 43, 373–383.
- 596 Tilman, D., Isbell, F., Cowles, J.M., 2014. Biodiversity and Ecosystem Functioning. *Annu. Rev. Ecol.*
597 *Evol. Syst.* 45, 471–493. <https://doi.org/10.1146/annurev-ecolsys-120213-091917>
- 598 Triviño, M., Pohjanmies, T., Mazziotta, A., Juutinen, A., Podkopaev, D., Le Tortorec, E., Mönkkönen,
599 M., 2017. Optimizing management to enhance multifunctionality in a boreal forest landscape. *J.*
600 *Appl. Ecol.* 54. <https://doi.org/10.1111/1365-2664.12790>
- 601 Tuomi, M., Laiho, R., Repo, A., Liski, J., 2011. Wood decomposition model for boreal forests. *Ecol.*
602 *Modell.* 222, 709–718.

603 Tuomi, M., Thum, T., Järvinen, H., Fronzek, S., Berg, B., Harmon, M., Trofymow, J.A., Sevanto, S.,
604 Liski, J., 2009. Leaf litter decomposition—Estimates of global variability based on Yasso07
605 model. *Ecol. Modell.* 220, 3362–3371.
606 <https://doi.org/http://dx.doi.org/10.1016/j.ecolmodel.2009.05.016>

607 Vaara, M., Saastamoinen, O., Turtiainen, M., 2013. Changes in wild berry picking in Finland between
608 1997 and 2011. *Scand. J. For. Res.* 28, 586–595.
609 <https://doi.org/10.1080/02827581.2013.786123>

610 van der Plas, F., Manning, P., Allan, E., Scherer-Lorenzen, M., Verheyen, K., Wirth, C., Zavala, M.A.,
611 Hector, A., Ampoorter, E., Baeten, L., Barbaro, L., Bauhus, J., Benavides, R., Benneter, A.,
612 Berthold, F., Bonal, D., Bouriaud, O., Bruelheide, H., Bussotti, F., Carnol, M., Castagneyrol, B.,
613 Charbonnier, Y., Coomes, D., Coppi, A., Bastias, C.C., Muhie Dawud, S., De Wandeler, H.,
614 Domisch, T., Finér, L., Gessler, A., Granier, A., Grossiord, C., Guyot, V., Hättenschwiler, S.,
615 Jactel, H., Jaroszewicz, B., Joly, F.-X., Jucker, T., Koricheva, J., Milligan, H., Müller, S., Muys,
616 B., Nguyen, D., Pollastrini, M., Raulund-Rasmussen, K., Selvi, F., Stenlid, J., Valladares, F.,
617 Vesterdal, L., Zieliński, D., Fischer, M., 2016. Jack-of-all-trades effects drive biodiversity-
618 ecosystem multifunctionality relationships in European forests. *Nat. Commun.* 7, 11109.
619 <https://doi.org/10.1038/ncomms11109>

620 Watson, J.E.M., Shanahan, D.F., Di Marco, M., Allan, J., Laurance, W.F., Sanderson, E.W., Mackey,
621 B., Venter, O., 2016. Catastrophic declines in wilderness areas undermine global environment
622 targets. *Curr. Biol.* 26, 2929–2934. <https://doi.org/10.1016/J.CUB.2016.08.049>

623 Wistbacka, R., Orell, M., Santangeli, A., 2018. The tragedy of the science-policy gap – Revised
624 legislation fails to protect an endangered species in a managed boreal landscape. *For. Ecol.*
625 *Manage.* 422, 172–178. <https://doi.org/10.1016/J.FORECO.2018.04.017>

626 Zubizarreta-Gerendiain, A., Pukkala, T., Peltola, H., 2019. Effect of wind damage on the habitat
627 suitability of saproxylic species in a boreal forest landscape. *J. For. Res.* 30, 879–889.
628 <https://doi.org/10.1007/s11676-018-0693-7>

629 Zubizarreta-Gerendiain, A., Pukkala, T., Peltola, H., 2016. Effects of wood harvesting and utilisation
630 policies on the carbon balance of forestry under changing climate: a Finnish case study. *For.*
631 *Policy Econ.* 62, 168–176. <https://doi.org/10.1016/J.FORPOL.2015.08.007>

632

633

634

635

		NPI constraint				No NPI constraint			
		ESS MF	CM MF	VH MF	DW MF	ESS MF	CM MF	VH MF	DW MF
All management regimes	MAX ESS MF	0.497	<u>0.150</u>	0.027	0.075	0.694	0.395	0.160	0.265
	MAX CM MF	0.354	0.272	<u>0.011</u>	0.063	0.528	0.995	0.234	<u>0.055</u>
	MAX VH MF	<u>0.345</u>	0.179	0.379	0.077	0.508	0.441	0.686	0.386
	MAX DW MF	0.372	0.182	0.122	0.171	0.416	0.586	0.238	0.618
CCF management regimes	MAX ESS MF	0.580	<u>0.164</u>	<u>0.034</u>	<u>0.100</u>	0.688	0.414	0.173	0.283
	MAX CM MF	0.481	0.212	0.037	0.117	0.529	0.994	0.238	0.054
	MAX VH MF	0.466	0.170	0.376	0.108	0.513	0.444	0.684	0.382
	MAX DW MF	0.482	0.179	0.124	0.168	<u>0.453</u>	0.575	0.347	0.591
RF management regimes	MAX ESS MF	0.389	0.171	0.060	0.075	0.563	0.689	0.225	0.136
	MAX CM MF	0.328	0.268	<u>0.042</u>	<u>0.046</u>	0.528	0.994	0.234	0.052
	MAX VH MF	<u>0.312</u>	0.149	0.182	0.095	0.528	0.935	0.270	0.086
	MAX DW MF	0.319	<u>0.136</u>	0.085	0.168	0.430	0.516	0.098	0.420

636

637 Table 1. Payoff table between component groups, for each of the component groups (ESS MF –
 638 Ecosystem service multifunctionality, CM MF – Climate mitigation multifunctionality, VH MF -
 639 Vertebrate habitat multifunctionality and DW MF – Deadwood habitat multifunctionality). Maximal
 640 values are bolded, while the minimal values are underlined.

641

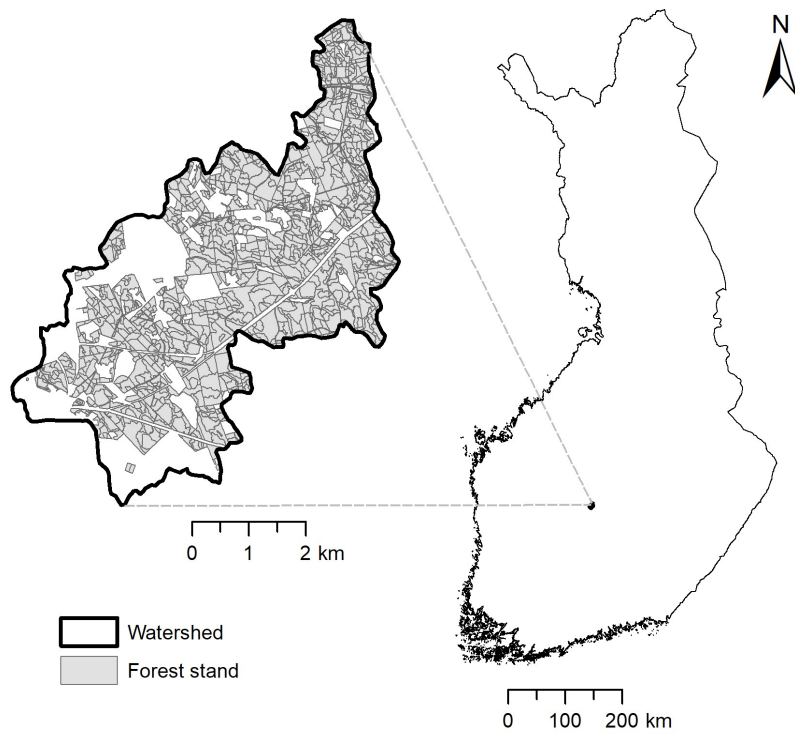
<u>Symbol</u>	<u>Definition</u>
Sets:	
B	Set of components
T_b	Set of criteria use in analysis, for each component b
P	Set of time periods under consideration
J	Set of all forest stands
K_j	Set of all management alternatives for forest stand j
Data:	
z_{jkp}^t	The value of criterion t when conducting management alternative k on stand j for period p
f_t^*	The ideal value obtainable for the criterion t
f_{t*}	The anti-ideal value obtainable for criterion t
D_b^*	The ideal value obtainable for the multifunctionality component b
D_{b*}	The anti-ideal value obtainable for the multifunctionality component b
Variables:	
D_b	The deviations away from the each component of multifunctionality
f_t	The value obtained for criterion t
Decision Variables:	
x_{jk}	The decision to manage stand j according to management alternative k
Parameters:	
r	The discount rate
q	Required proportion of maximum net present value

642

643 Table 2. A list of notations used throughout the paper.

644

645

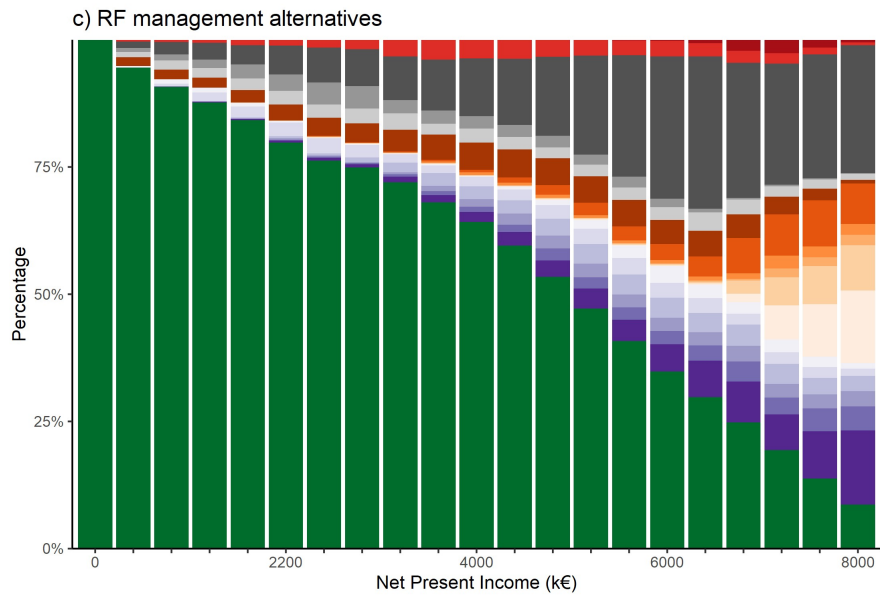
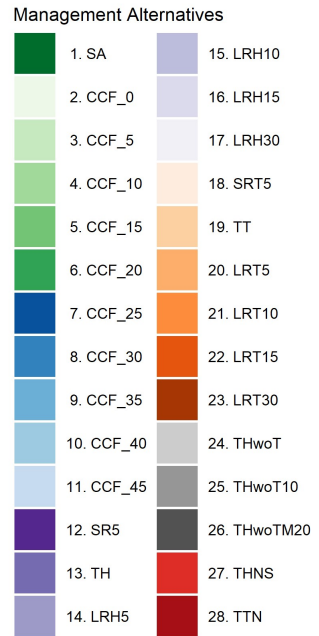
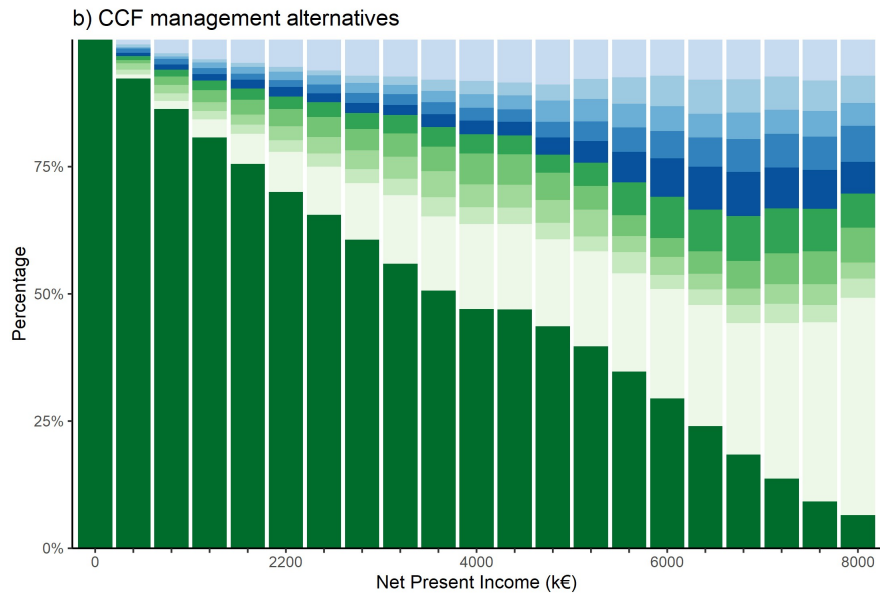
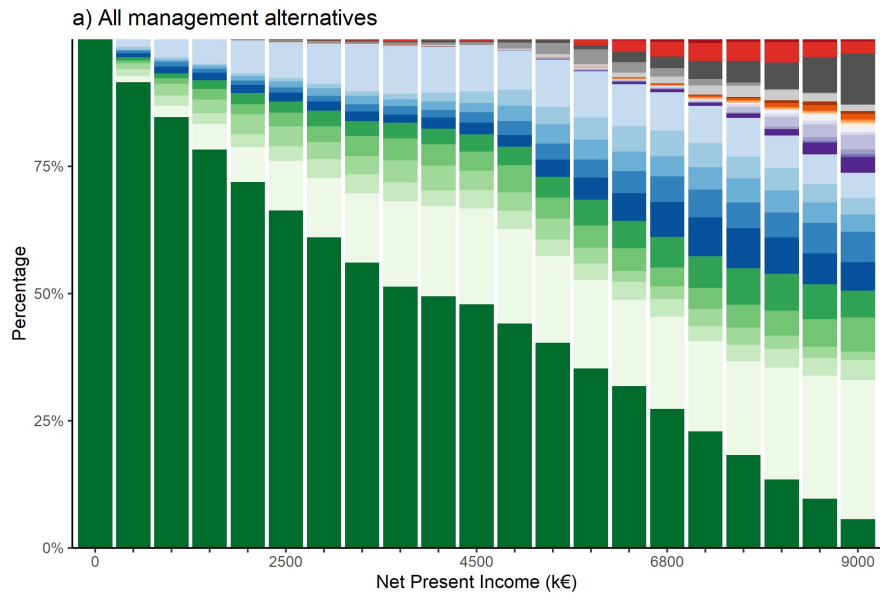


646

647 Fig. 1. Location of the forested watershed in Central Finland and location of individual forest stands.

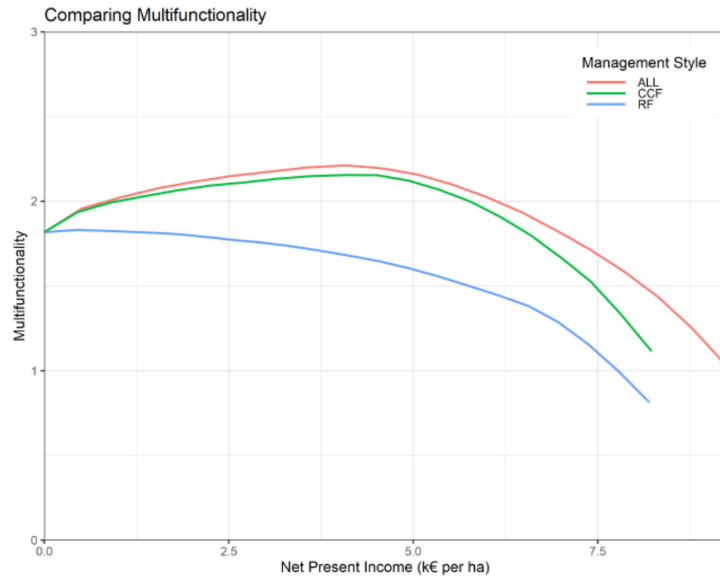
648

649

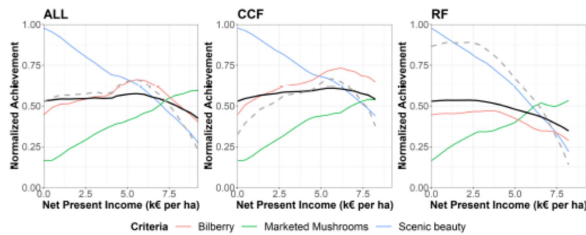


651 Fig. 2. Land-use intensity in terms of net present income of the different scenarios, measured as
652 proportion of unmanaged forests. Between the figures the x-axes have slightly different range, as
653 each scenario has differing maximal values. All – all management options allowed, CCF – only
654 continuous cover forest (alternatives 1 – 11), RF – only rotation forestry management (alternatives 1
655 and 12-28). Note: to aid in figure clarity, the modifications to the BA requirement for CCF harvesting
656 are aggregated and represents a total of 40 alternatives. For a detailed explanation of the
657 management alternatives, readers are guided to Appendix A.

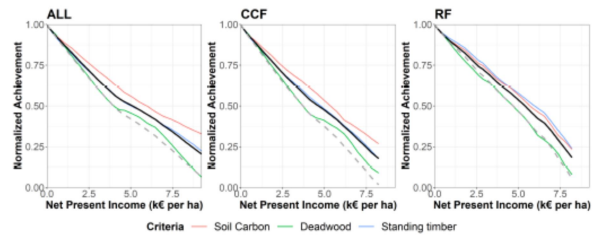
658



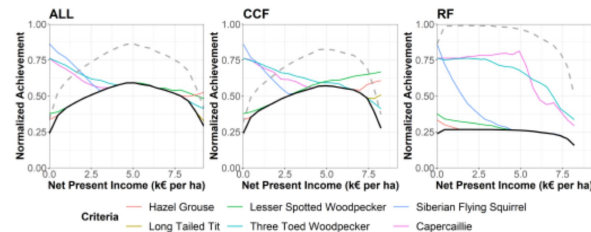
(a) landscape level multi-functionality



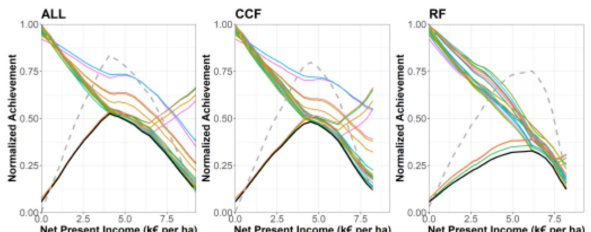
(b) Ecosystem service multi-functionality



(c) Climate mitigation multi-functionality



(d) vertebrate habitat multi-functionality



(e) Deadwood habitat multi-functionality

660
661

662 Fig. 3. Comparison of the multifunctionality measures for a) landscape-scale multifunctionality, and
 663 b)-e) individual multifunctionality components and their indicators. The black line represents the
 664 distance value for the set of indicators of a specific component: average value (b, c), or minimum
 665 value across indicators (d, e). The grey dashed line represents the normalized distance value from
 666 the range within each component groups (scaled with the minimal and maximum values from the
 667 payoff table (Table 1)). The list of names for the 32 deadwood habitats can be found in Tikkanen et

668 al. 2007. ALL - All management alternatives are allowed, CCF - only continuous cover forestry
669 alternatives are allowed, RF - only rotation forestry alternatives are allowed.