

1 *Original research article*

2 **Conflicting interests of ecosystem services: multi-criteria modelling and indirect**
3 **evaluation to trade off monetary and non-monetary measures**

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9

10 **Abstract**

11 Ecosystems provide services for many stakeholder groups, often with a conflict of interests that
12 hampers sustainability. Core to these conflicts is the challenge of trading-off monetary and non-
13 monetary measures. Using the boreal forest as a case, we present a socio-ecologically integrated trade-
14 off model for partly competing services (wood, game hunting, livestock grazing). Drawing on multi-
15 criteria analyses (MCA), we found that wood production unequivocally yielded the highest net present
16 value, but led to a substantial reduction in the performance of hunting and grazing. By imposing
17 multiuse conditions set as minimum performance of the less profitable services, we evaluated the
18 opportunity costs of multiuse without directly pricing non-commodities. We also quantified
19 normalized indices of realized performance potential to evaluate the cost of multiuse with a single,
20 joint metric. Both approaches consistently showed that accepting a rather small loss in one service
21 may secure large gains in other services. By democratically providing a combined monetary and non-
22 monetary evaluation, our approach should facilitate broader acceptance for the decisional metrics
23 among stakeholders. It thereby has the potential to mitigate conflicts, feeding into the larger scheme of
24 adaptive management.

25

26 **Key-words:** bioeconomy; bio-socio-economy; logging; MCDA; multi-use; optimization

27 **1 Introduction**

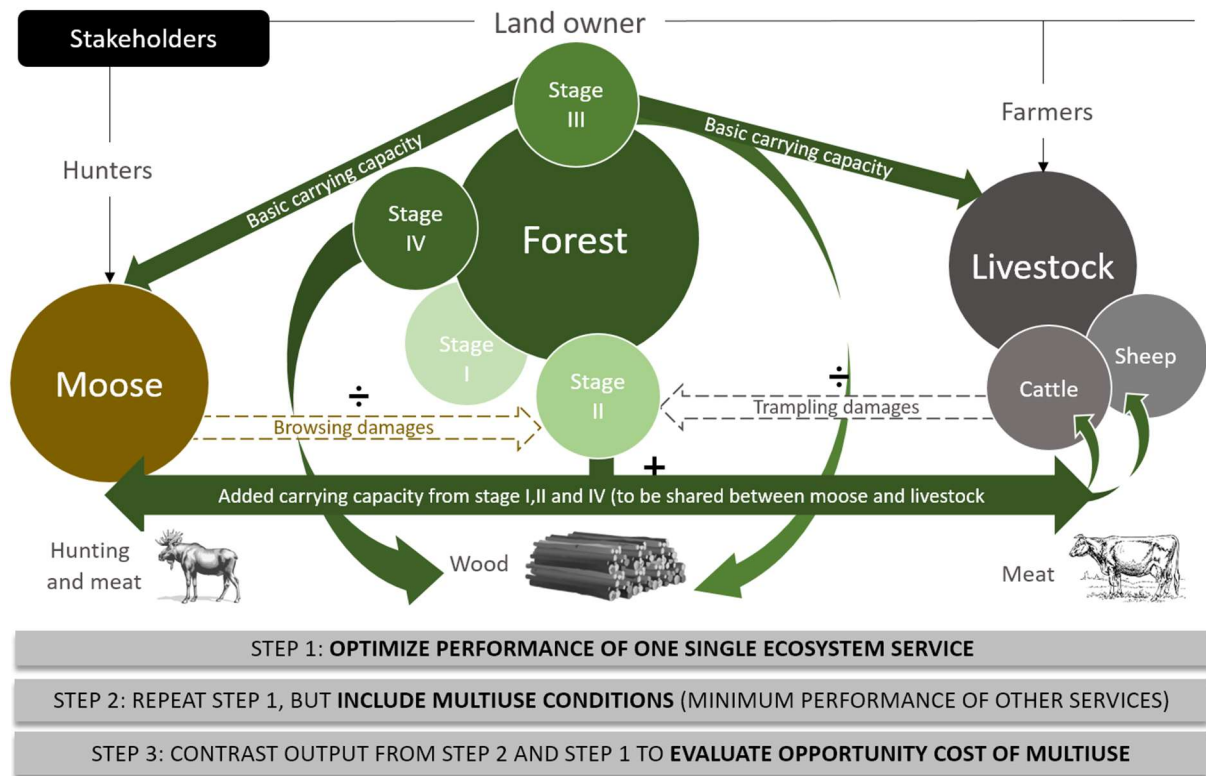
28 With a steadily rising human population and increasing needs for renewable resources, policymaking
29 for ecosystem services is more challenging than ever (Lindenmayer et al. 2012). Such intensification
30 of pressures on resources raises the potential for conflict between stakeholder interests, because most
31 ecosystems are utilized for different and competing services (de Groot et al. 2010). This is
32 counterproductive to sustainability, given that conflicts exacerbate overexploitation (*sensu* the tragedy
33 of the commons, Hardin 1968) (Redpath et al. 2015). In some cases conflicts may be socially
34 productive by disrupting skewed distribution of benefits (Tjosvold 1991). More typically, however,
35 conflicts also hamper socioeconomic value creation (Arancibia 2013; Hotte 2001), a proclaimed goal
36 of many nations around the globe (Bioeconomy Council 2013; OECD 2009).

37 Our ability to solve these conflicts is limited by a lack of scientific approaches that can aid in
38 comprehensively identifying the optimal management strategy when stakeholder interests clash
39 (Maxwell et al. 2014; Redpath et al. 2013). There is broad consensus that incorporating the views of
40 all interest groups is essential for managing conflicts (e.g., Dennis et al. 2005; Kyllönen et al. 2006).
41 With ecosystem services, comprehensive approaches typically must involve trading off multiple
42 interests (Rodríguez et al. 2006, 2012), adding complexity to the challenge. At the heart of these
43 shortcomings is a persistent dichotomy between monetary and non-monetary goals, and the inherent
44 difficulties of finding joint decision metrics that the opposing parties can agree upon (Wam 2010).

45 How and whether we should evaluate non-marketable ecosystem services is no small debate.
46 Alternative currencies have been put forward, such as energy (McKibben 2007) or happiness
47 (MacKerron 2012), but the decisional power remains in the favour of interests operating in monetary
48 markets (Adamowicz 2004). Non-monetary measures are nevertheless imperative to the sustainable
49 use of ecosystem services as the limits ultimately is biophysical, not economic (Fischer et al. 2007).
50 Advancement of ways to calculate and combine decision metrics in trade-off protocols is therefore
51 gaining research focus (Diaz-Balteiro & Romero 2008; Ostrom 2007; Schlüter et al. 2014). Poff et al.
52 (2010), for example, illustrate a most comprehensive use of compromise programming to aid multi-
53 criteria decision planning by simultaneously optimizing multiple objectives (e.g., plant productivity,
54 biodiversity, streamflow rates, habitat suitability and willingness-to-pay for recreation opportunities).
55 This much-aspired inclusiveness comes with a cost of immense trade-off complexity, which forces

56 that we measure service performances by some kind of normalized indices. Planning participants
57 typically find it difficult to interpret such relative indices (Kangas et al. 2001), and prefer to base their
58 decisions on hands-on measures like biomass or money (but see Adamowicz 2004, p. 439). Along
59 with the ongoing and promising development of multi-criteria analysis (collectively labelled MCA),
60 we advocate to simultaneously explore other ways of implementing trade-off assessment without
61 direct pricing, yet within the ruling scheme of monetary exchange protocols (for a recent review of
62 established and suggested such approaches, see Schuhmann & Mahon 2015).

63 Aiming at socio-ecological integration, we outline a dynamic trade-off model for the optimization
64 of ecosystem services with partly conflicting stakeholder interests, when land sharing is the preferred
65 option. The inclusion of non-monetary goals and concerns adds new dimensions to the underlying
66 traditional Pareto optimization. Drawing on goal programming (Tamiz et al. 1998), we made factorial
67 comparisons of both monetary and non-monetary output from scenarios with contrasting service
68 priorities. By imposing multiuse conditions set as minimum performance of the less profitable
69 services, we evaluated the opportunity costs of multiuse without direct pricing of the non-commodities
70 (Fig. 1). Drawing also on elements from compromise programming (Zeleny 1974), we additionally
71 quantified normalized indices of realized performance potential to evaluate the cost of multiuse with a
72 single, joint measure. By democratically providing a comprehensive monetary and non-monetary
73 evaluation, our approach should generate broader stakeholder acceptance for the decisional metrics
74 (Ostrom 2007; Milner-Gulland 2011). It thereby has the potential to mitigate conflicts, feeding into the
75 larger schemes of adaptive management, such as the management strategy evaluation (Mapstone et al.
76 2008) or multi-criteria decision support (Kangas & Kangas 2005).



77

78 **Figure 1.** The use of one ecosystem service may both impede and facilitate other services, as partly
 79 illustrated above using forest as a case: wood logging in older forest (stage III-IV) substantially
 80 contributes to food carrying capacity for moose and livestock, but livestock cause trampling damages
 81 and moose cause browsing damage to the new recruitment of trees (stage I-II). In our trade-off model,
 82 we sequentially assess the effects of favouring single or all stakeholder groups on not only monetary
 83 output (net present value), but also goods and services (hunting, wood and meat). Because different
 84 stakeholder groups have different goals and gains, also of non-economic value, trading-off the
 85 conflicting services using only a monetary measure is likely to exacerbate conflict.

86 **2 Model framework**

87 *2.1 Model objectives*

88 We used the Nordic boreal forest as a case study, with three partly competing services: wood
89 production, game hunting (moose *Alces alces*) and livestock grazing (sheep *Ovis aries*, cattle *Bos*
90 *taurus*.) Here we test four scenarios with contrasting objective functions: (1) prioritize wood
91 production (WOOD), (2) prioritize game hunting (HUNT), (3) prioritize livestock grazing (GRAZ),
92 and (4) prioritize multiuse: i.e. maximize total performance given various levels of multiuse conditions
93 (TRI-0 = no such conditions, TRI-L = low levels, TRI-H = high levels). The TRI-L and TRI-H
94 represent non-Pareto solutions, where we imposed conditions as minimum performance of less-
95 profitable services (see also Fig. 4 for additional multiuse levels).

96 We ran the model as a non-linear numerical optimization problem (NLP) in GAMS (20.7,
97 Windows NT) using the CONOPT3[®] solver (Drud 2006). We first solved our objective function by
98 applying a maximization statement on the net present value equation of interest (eq. 1-4, depending on
99 the ecosystem service to be prioritized). As an alternative to these objective functions based on net
100 present value, we also optimized the model using normalized indices of realized performance potential
101 (eq. 7). Here we applied a parallel to the approach used in compromise programming of minimizing
102 the distance to an ideal, but unattainable point (Zeleny 1974). By minimizing the sum of these
103 distances across all three ecosystem services, we could further explore the effects of multiuse by
104 assigning equal or different weights to each service. Different weighting of services may be crucial in
105 the final decision process when non-commodities are involved (Hajkowicz 2008).

106

107 *2.2 Model structure*

108 To facilitate readability we have kept most of the mathematics in the supplementary appendix. In the
109 following equations with an A in front refers to this appendix. The growth of both tree and animal
110 populations were modelled with a stage-structured version (Usher 1966, 1969) of basic Leslie matrices
111 (Leslie 1945) (eq. A1-A6). The model is projected at one-year intervals over a finite planning period,
112 assuming discrete reproduction and mortality. Reflecting what is recognizable for the hunters, the
113 moose population M_t consists of five stages (calves, female or male yearlings, older cows or bulls).
114 The cattle population C_t consists of four stages (female or male calves, female heifers, older cows).

115 The sheep population S_t has only three stages as sheep give birth as yearlings (female or male lambs,
116 older ewes). Livestock males 1+ years old are not allowed on forest pastures, so their survival is set to
117 zero. In the model, they must therefore be slaughtered in their first year of life to generate income.

118 The forest is divided into strata comprising two variables: the tree species of commercial interest
119 (Norway spruce *Picea abies*, Scots pine *Pinus silvestris* and birch *Betula* spp.), and the site's innate
120 capacity to produce forest (hereafter termed Site Index: low ($H_{40} = 7-11$), intermediate ($H_{40} = 14-17$)
121 and high ($H_{40} = 21$) (see Tveite 1977). For each stratum we have four tree stages: I = trees covered by
122 snow in winter and unavailable to foraging animals (tree height 0.0–0.3 m), II = trees with major parts
123 of their crown within all-year reach of foraging animals (tree height 0.3–3.0 m), III and IV = trees with
124 their crowns fully above the reach of foraging animals. Average age intervals of stages are given in the
125 supplementary appendix, [Table A.1](#). Only trees in stages III and IV have market value. New trees are
126 always recruited after harvest, and only to stage I. We assume that all logging is undertaken as clear-
127 felling (an important assumption when calculating costs and animal carrying capacity).

128 Density dependent ungulate-forest interactions are included in the model by adding a non-linear
129 function to the population projections (eq. [A7](#)). We base these functions on logistic growth, so that the
130 effect is less intense initially, and then increases before levelling off towards carrying capacity
131 saturation (eq. [A8](#)). The forest's capacity to sustain foraging ungulates (denoted K_m , K_s and K_c for
132 moose, sheep and cattle respectively) consists of two parts (eq. [A9](#)). One is the basic carrying capacity,
133 defined as the number of animals sustained when the entire forest is in the least forage producing stage
134 (stage III). The other part is added capacity from forest stages other than stage III. Recently logged
135 sites (stage II) are of particular importance, because of their much higher forage abundance. The added
136 capacity for each stage varies with tree stratum and animal species. For example, stage I (field layer
137 dominated by grass) is of higher value to cattle than to moose, while stage IV (field layer dominated
138 by bilberry) is of higher value to moose than to cattle.

139 Hunted moose ($h_{t,k}$) and slaughtered livestock ($sc_{t,k}$, $ss_{t,k}$) generate a monetary value (pm , pc , ps)
140 (€) paid per kilo of meat (dressed carcass weight $w_{m,k}$, $w_{c,k}$, $w_{s,k}$). For moose, there is also a fixed stage-
141 specific hunting fee paid per animal hunted (ph_k), irrespective of body mass. Total net present value of
142 moose, cattle and sheep (π_m , π_c , π_s , respectively) (€) is:

$$143 \quad \pi m = \sum_{t=1}^T \sum_{k=1}^K \delta^t \cdot \left[ph_k + pm \cdot wm_k \cdot \left[1 + \eta_k \cdot (M_t / Km_t)^{\rho_k} \right]^{-1} \right] \cdot h_{t,k} + MEV \quad (1)$$

$$144 \quad \pi c = \sum_{t=1}^T \sum_{k=1}^K \delta^t \cdot \left[pdays / 365 \cdot pc \cdot wc_k \cdot \left[1 + \eta_k \cdot (C_t / Kc_t)^{\rho_k} \right]^{-1} \right] \cdot sc_{t,k} + CEV \quad (2)$$

$$145 \quad \pi s = \sum_{t=1}^T \sum_{k=1}^K \delta^t \cdot \left[pdays / 365 \cdot ps \cdot ws_k \cdot \left[1 + \eta_k \cdot (S_t / Ks_t)^{\rho_k} \right]^{-1} \right] \cdot ss_{t,k} + SEV \quad (3)$$

146 where δ^t is the discount factor, which is included because future income is associated with uncertainty
 147 (for a discussion of the dilemmas of discounting, see Philibert 2003) and $pdays$ are the number of days
 148 in the forest pasturing season (reflecting that livestock income does not only stem from forest
 149 pasturing). The species-specific constants η_k and ρ_k adjust the density influence on animal body mass
 150 (influence being stronger for sub-adults). As a rule of thumb, boreal forest plants can sustain a
 151 browsing intensity which removes about 1/3 of their current growth (Speed et al. 2013). Therefore, η_k
 152 and ρ_k are set to reduce body mass fairly slowly until M_t/Km_t is about 1/3, then intensifying before
 153 levelling off when M_t/Km_t reaches about 2/3, reflecting that foraging will be increasingly energy costly
 154 to obtain as tree growth and the available biomass/tree declines. MEV , CEV and SEV in eq. 1-3 are
 155 expectation values, included to avoid complete decimation of the populations at the end of the
 156 planning period (see eq. A12 in supplementary appendix).

157 Trees are harvested at various stages in each stratum. The total net present value (πf) is:

$$158 \quad \pi f = \sum_{t=1}^T \sum_{s=1}^S \delta^t \cdot (pf_s \cdot u_{t,s} - cf_s - af - cr_s - cM_{t,s} - cC_{t,s}) + FEV \quad (4)$$

159 where pf_s is the net revenue (harvesting costs deducted) (€) per m³ of wood cut in stratum s , $u_{t,s}$ is the
 160 amount of wood (m³) cut at time t (volumes of trees are stage-specific for a given stratum), cf_s is the
 161 fixed cost of conducting one cutting session (e.g., costs of moving equipment between sites, or pre-
 162 cutting surveys). Because our model is not spatially explicit, we have to assume that all cutting within
 163 a stratum-specific stage represents one cutting session (thus if a stratum is cut in a given year, one unit
 164 of cf_s will be deducted). af is the fixed administrative cost of managing the forest. The latter is
 165 deducted from the wood income (rather than game or livestock) as forestry normally is the focal
 166 interest of landowners in Nordic boreal forests. Forest recruitment after cutting is associated with a
 167 cost in spruce forest cr_s (i.e. planting of nursery grown saplings, eq. A11), but not in pine or birch
 168 forest (which are recruited by natural seeding). FEV is the forest expectation value (see eq. A10):

169 In eq. 4, cM_t and cC_t are the costs of having moose and cattle in the forest, in terms of browsing
 170 damage on pines in stage II (moose), and trampling damage on spruce and birch in stages I-II (cattle).
 171 In this study, moose is not considered to cause commercial damage to birch or spruce. Only pines in
 172 stage II are damaged by moose browsing, because trees in stage I are covered by snow in winter (pine
 173 is winter forage for moose). Trampling damage does not pertain to pine as pine clear-cuts do not have
 174 the intense upsurge of grass coverage that cattle are seeking. In this study, sheep are not considered to
 175 damage any of the tree species of commercial interest (Hjeljord et al. 2014). All damage depends on
 176 animal density and carrying capacity at the time:

$$177 \quad cM_{t,s} = \delta^{T_H} \cdot \bar{p}f \cdot \psi_s \cdot f_{t,s} \cdot \sum_{k=1}^K (M_{t,k} \cdot b_k) / Km_t \cdot (1 + \alpha^{\beta \cdot M_t \cdot Km_t^{-1}})^{-1}, \quad s \in \{pine, k = II\} \quad (5)$$

$$178 \quad cC_{t,s} = \delta^{T_H} \cdot \bar{p}f \cdot \psi_s \cdot f_{t,s} \cdot \theta \cdot pdays C_t \cdot (f_{t,s} / td_{t,s})^{-1}, \quad s \in \{spruce, birch, k = I, II\} \quad (6)$$

179 where δ^{T_H} is the discount factor T_H years in time, which corresponds to the time it takes for the average
 180 tree of stage II to reach the midpoint between stages III and IV. The monetary value of this tree ($\bar{p}f$) is
 181 calculated as the average profit of a tree cut in stage III–IV across the strata of interest.

182 In eq. 5, the constant b_k adjusts the browsing influence of different moose stages (adults are
 183 browsing more trees than sub-adults). The proportion of pines that will be browsed increases linearly
 184 with moose density in relation to carrying capacity. The two constants α and β regulate the severity of
 185 browsing damage (i.e. the proportion of browsed trees that will lose all monetary value); it will be
 186 higher when the moose population is closer to its carrying capacity, as browsing per tree then
 187 intensifies and more trees will reach their browsing resilience limit. Because moose typically first aims
 188 at the leader shoot, which is crucial for the growth and quality of pine timber, α and β are set so that at
 189 least 50% of browsed pines will be damaged even at low moose densities. The cost of damaged pine is
 190 corrected with a stem thinning factor ψ_s (tree density at midpoint stage III and IV / tree density at stage
 191 II) to take into account that even without moose damage, the tree density decreases with time.

192 In eq. 6, the constant θ is the proportion of new spruce saplings that is trampled each year per
 193 cattle-day in the forest. All cattle (cows, heifers and sucklings) are considered to make similar levels
 194 of trampling damage. Because even minor trampling damage incurs a severe reduction in future timber
 195 quality of spruce, all damaged saplings lose all their monetary value. The proportion of trampled
 196 saplings increases both with more cattle-days or with lower proportions of the forest being in stages I

197 and II. The latter occurs because more cattle will then aggregate in these areas, as clearcuts are highly
198 selected habitat for cattle. As for browsed pine, the cost of damaged spruce is corrected with a
199 thinning factor ψ_s (tree density at midpoint stages III and IV / tree density at stage I).

200 We also calculated normalized indices of realized performance potential. For hunting (H) and
201 grazing (C and S) the performances were measured in terms of kilos meat produced throughout the
202 planning period. For wood production (F), the potential was measured in terms of net present value
203 stemming from timber. The normalized indices of each were summed to obtain a single maximization
204 metric (I) encompassing all three ecosystem services:

$$205 \quad I = (w_h \cdot H^* / H_{\max} + w_c \cdot C^* / C_{\max} + w_s \cdot S^* / S_{\max} + w_f \cdot F^* / F_{\max}) / \sum_{i=1}^k w_i \quad (7)$$

206 where H_{\max} , C_{\max} , S_{\max} and F_{\max} are the potentials as found by maximizing each performance in
207 individual model runs, H^* , C^* , S^* and F^* are the performances to be jointly maximized through the
208 use of I , and w_i are weighting factors to prioritize ecosystem service i in relation to the other services.
209 Each of the performance fractions (e.g., H^*/H_{\max}) as well as the joint metric I becomes a relative scale
210 0-1, where 1 = maximum potential realized.

211

212 2.3 Model constraints set by non-commodity concerns

213 Not all elements of the forest ecosystem can be adequately addressed with economic theory (Wam
214 2010). We set the following non-commodity concerns as model constraints (their effect on economic
215 and biological output is addressed in our previous work, Wam & Hofstad 2007).

- 216 (i) In line with the ethical notion of sustainability (Leopold 1949), all animal populations must
217 remain below their specific carrying capacity at all times.
- 218 (ii) Moose fecundity (as influenced by animal density) must stay ≥ 0.5 calves produced per cow 2+
219 years. Lower values indicate severe deterioration of health (Solberg et al. 2006). No constraint is
220 set for livestock as their fecundity is determined *ex-situ* by the farming regime, and treated as a
221 constant in the model (Table A.1).
- 222 (iii) In line with perceived hunter ethics, moose calves cannot be orphaned by hunters, i.e. the number
223 of hunted cows must not exceed the number of hunted calves divided by the live calf: cow ratio.

224 (iv) The moose cow: bull ratio must stay ≤ 1.8 to secure breeding conditions and to avoid delayed
225 parturition (Sæther et al. 2003) or skewed sex-ratios of new-borns (Sæther et al. 2004).

226

227 *2.4 Model parameterization and parameter sensitivity*

228 To illustrate the model we used a 67 000 ha large forest (43 000 ha productive land) with baseline
229 conditions set to resemble contemporary market values and activity levels in the Nordic countries
230 (Table A.1-A.2). Most ecosystem services in the Nordic forests are loosely regulated by public law,
231 and in practice managed by the landowner (private citizens, commons or companies). The landowner
232 typically decides about forest harvesting and moose hunting, but often have less influence on the
233 intensity of livestock grazing (Berge 2002). For example, grazing rights may stem from a time where
234 subsistence and not commercial interests were the prevailing driver, and thus is not quantitatively
235 limited in modern terms. Informal institutions also influence decision-making: moose hunting, for
236 example, is a club good with strong cultural ties to local hunters (Jacobsen 2014). If the landowner
237 prioritizes wood harvest at the expense of hunting or grazing, he may lose goodwill in the community.

238 Forest growth, moose demography and in part moose: forest interactions were parameterized and
239 empirically validated in our earlier work (Wam & Hofstad 2007). The model was updated with new
240 field data on moose-forest interactions (Wam & Hjeljord 2010; Wam et al. 2010). We collected data
241 on livestock demography from the Norwegian Agriculture Agency, and cattle trampling damage from
242 own field studies (Hjeljord et al. 2014). Livestock habitat use and diet in forests, and their niche
243 overlap with moose were obtained by conducting new field work (Wam, unpublished data).

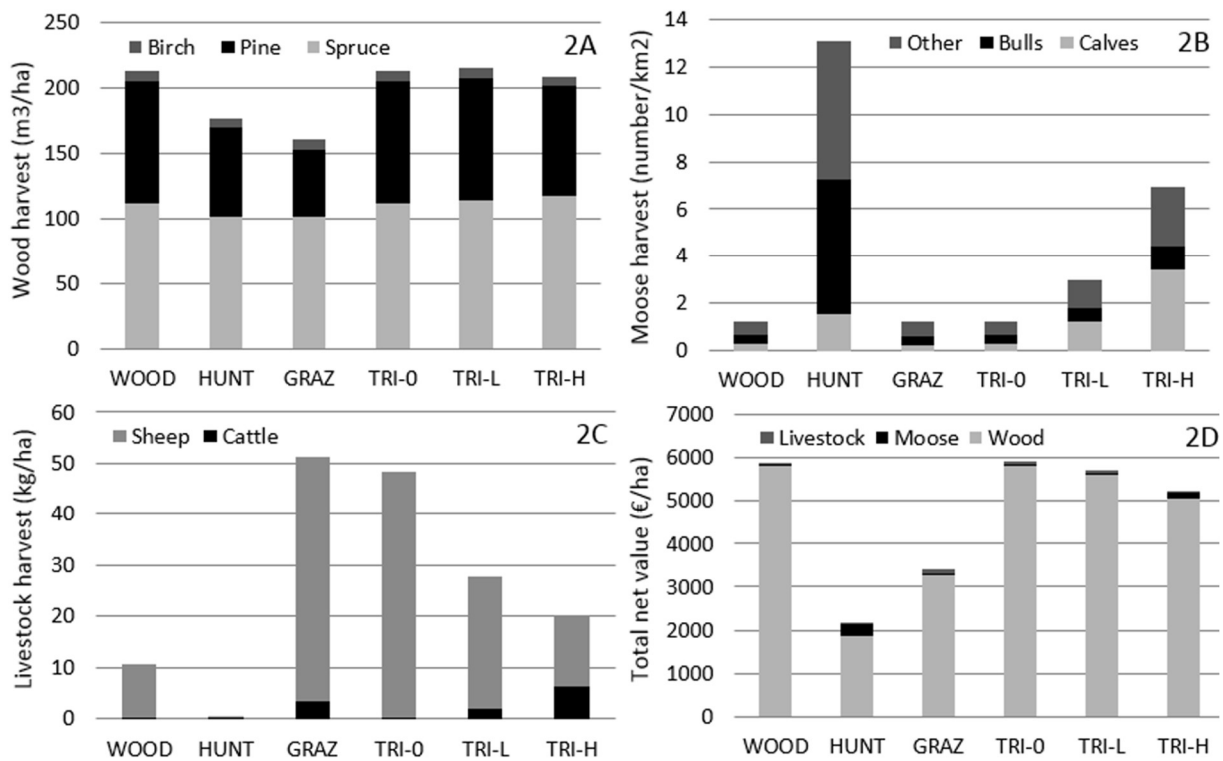
244 The planning period was set to 30 years, and the interest rate to 3%. These factors will influence
245 the level of generated net present value, but negligibly affect the relative contribution of wood versus
246 game or livestock when all resources are assigned expectation values (see also **Table 1**). All constant
247 or initial parameter values used in the model are given in **Tables A.1 and A.2**. We inferred parameter
248 sensitivity by successively rerunning the model while rescaling one parameter at a time. Due to the
249 many parameters, we mostly report output for three input levels: contemporary settings (hereafter
250 called baseline), a realistic lower extreme and a realistic upper extreme. For parameters with patterns
251 of particular interest we also report selected output on a more continuous scales.

252 **3 Results**

253 **3.1 Prioritizing wood production (WOOD)**

254 Wood had about 2-3 times higher income potential than hunting and grazing (Fig. 2D), making it
 255 financially beneficial to minimize browsing and trampling damage. The optimal strategy both when
 256 maximizing net present value of wood (WOOD) and when maximizing total net present value (TRI-0),
 257 was therefore to eliminate moose and cattle, while keeping sheep at moderate densities (Fig. 2B-C). In
 258 the WOOD scenario, wood consistently contributed 98-99% of the total net present value over time,
 259 for the whole range of applied parameter settings (Table A.2). Factors facilitating contribution of
 260 wood to the total net present value (W%) were: a higher market value of timber, a higher Site Index
 261 (i.e. more productive forest land), and more pine in the forest. With all these facilitating factors
 262 combined, the WOOD scenario could generate a mean annual net value from wood production of 885
 263 €/ha (compared to 215 €/ha with parameters set at baseline).

264



265

266 **Fig. 2.** Potential performance (A-C) and total net present value (D) of forest ecosystem services over 30 years
 267 according to a socio-ecologically integrated trade-off model for partly conflicting services, with the objective to
 268 maximize net present value from wood production (WOOD), game hunting (HUNT), livestock grazing (GRAZ),
 269 or total net present value given various levels of multiuse conditions. TRI-0 = no such conditions; TRI-L = low
 270 levels (at least 50 moose hunted, 100 cattle and 1 000 sheep pastured each year; TRI-H = higher levels (at least
 271 150 moose, 300 cattle and 3 000 sheep). Illustrated for a land area of 67 000 ha (43 000 ha productive forest).

272 3.2 Prioritizing game hunting (HUNT)

273 The optimal strategy when prioritizing game hunting (HUNT) was to eliminate all livestock (Fig. 2C),
274 maintain spruce harvest and reduce pine harvest (Fig. 2A). Hunting contributed a highly variable share
275 of the total net present value, depending on parameter settings (Table A.2). Factors facilitating the
276 contribution of hunting (H%) to the total net present value were: a higher hunting revenue (more so for
277 fees paid per-kilo than per-capita), a higher carrying capacity, a lower Site Index, more pine in the
278 forest, and higher damage intensity on browsed pines. With all these facilitating factors combined, the
279 HUNT scenario could generate a mean annual net value from moose hunting of 100 €/ha (compared to
280 15 €/ha with parameters set at baseline), i.e. only a fraction of the potential from wood production.

281 While the wood harvest (m³/ha) did not differ a lot between the HUNT and the WOOD scenarios,
282 the timber was logged at an earlier stage, facilitating shorter rotation times and larger areas being in
283 the more forage-productive younger stages. This and other (kb_m or ε_s , Table A.2) improvements of the
284 carrying capacity barely affected the total net present value, but greatly influenced the hunting
285 opportunities. The number of moose harvested in the HUNT scenario was ten times higher than in the
286 scenarios where moose was not explicitly prioritized (i.e. WOOD, TRI-0 and GRAZ) (Fig. 2B). Also,
287 a higher proportion of male moose (a target preferred by many hunters) was kept in the population as
288 well as harvested in the HUNT scenario compared to other scenarios.

289

290 3.3 Prioritizing of livestock grazing (GRAZ)

291 The optimal strategy when prioritizing livestock grazing (GRAZ) was to eliminate moose (Fig. 2B),
292 maintain the spruce harvest and reduce the pine harvest (Fig. 2A). Livestock had a generally low share
293 of the total net present value potential (Table A.2). Factors facilitating the relative contribution of
294 livestock (G%) to the total net present value were: a higher meat revenue, a higher carrying capacity, a
295 lower Site Index, and higher trampling intensity. Recall that spruce clearcuts were both the main
296 contributor to livestock carrying capacity and subject to livestock trampling damage. Consequently,
297 there were points of inflection in the influence of spruce proportion on livestock relative contribution
298 to net present value (being lower at intermediate spruce dominance). Sheep had a higher income (and
299 meat yield, Fig. 2C) potential than cattle. With all facilitating factors combined, the GRAZ scenario

300 could generate a mean annual net value from sheep of 40 €/ha and 8 €/ha for cattle, compared to 4
301 €/ha and 3 €/ha with parameters at baseline (sheep and cattle prioritized in separate model runs).

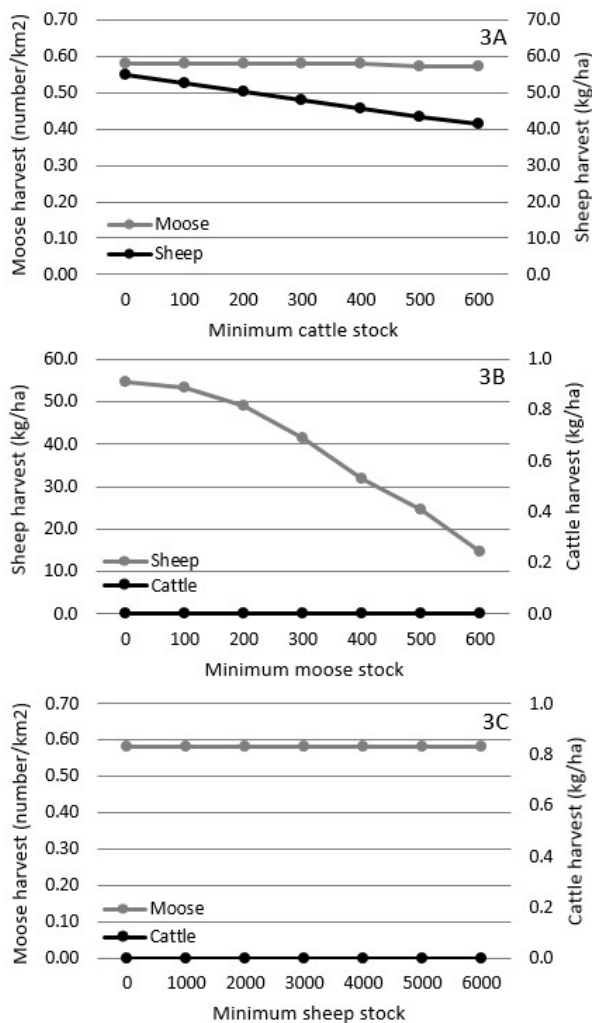
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303 3.4 Evaluating the opportunity cost of multiuse using minimum performance conditions (TRI-0, TRI-L, TRI-H)

304 Because of the superior income potential of wood, the TRI-0 scenario (i.e. maximizing total net value
305 without multiuse conditions) essentially gave the same performance as the WOOD scenario. The only
306 factor with noticeable influence on the relative contribution of the various ecosystem services was
307 very high revenues from animal meat (Table 1). Livestock grazing consistently had a marginally
308 higher contribution than moose hunting due to the lack of damage costs associated with sheep. The
309 TRI-H scenario (higher levels of multiuse conditions) involved a 12%, and the lower level scenario
310 TRI-L a 4%, reduction in total net present value compared to TRI-0.

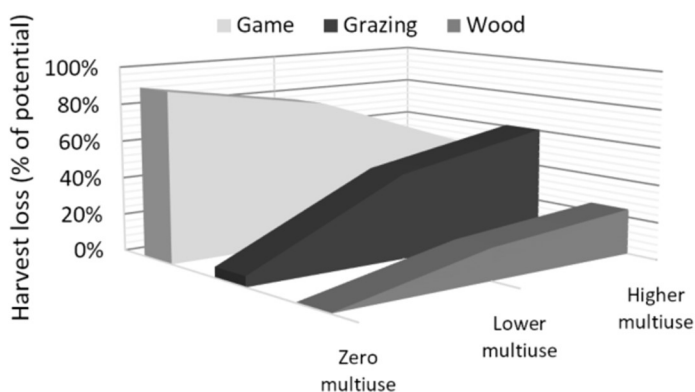
311 Compared to its effect on total net present value, adding multiuse conditions to the model more
312 strongly affected the biological output in terms of meat produced and game hunted. Raising the
313 minimum number of cattle in the forest had negligible influence on moose because of their low niche
314 overlap. The forced increase in cattle density was therefore countered in the optimization by a
315 reduction in the sheep density (Fig. 3A), in order to maintain low damage costs (i.e. a lowest possible
316 ratio of cattle equivalents to forest area in stage I-II, eq. 6). A forced increase in the minimum number
317 of moose in the forest was also countered by a reduction in sheep (Fig. 3B), as sheep and moose have
318 a higher niche overlap than cattle and moose (Table A.1). Raising the minimum number of sheep
319 allowed in the forest, on the other hand, did not influence the optimal density of either cattle or moose
320 (Fig. 3C), as the optimal sheep density without multiuse conditions (i.e. about 20 000 animals) anyway
321 superseded the levels we had set as minimum.

322 In contrast, raising the multiuse conditions to higher levels (TRI-H) generated a more fair
323 distribution of harvest loss (Fig. 4), still without jeopardizing much of the total net present value (see
324 Fig. 2D). Without multiuse conditions (TRI-0), game hunters carried practically all the burden of
325 being a less profitable stakeholder group. In TRI-0, their harvest was down by 90% compared to when
326 game hunting was prioritized. The wood production, on the other hand, was down by only about 20%
327 even with the higher multiuse conditions (TRI-H).



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Fig. 3. Potential performance of forest ecosystem services over 30 years according to a socio-ecologically integrated trade-off model for partly conflicting services (wood production, moose hunting and livestock grazing), with the objective to maximize total net present value given various levels of multiuse conditions, i.e. minimum performance of the monetarily less profitable services A) cattle, B) moose, and C) sheep (profit of wood production was superior to that of moose and livestock, thus not favoured with multiuse conditions).



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Fig. 4. Loss of potential performance from forest ecosystem services according to a socio-ecologically integrated trade-off model for partly conflicting services (wood production, moose hunting and livestock grazing), with the objective to maximize total net present value given three levels of multiuse conditions imposed to secure minimum performance of the monetarily less profitable services (i.e. grazing and game). The harvest potential (number of moose/km², kg livestock meat/ha or m³ of timber/ha) was calculated for a 30 year planning period, and equals the performance obtained if the ecosystem service in question was completely prioritized (i.e. maximizing the value of this service rather than the total value).

343 **Table 1.** Varying parameter values in an optimization model for management of forests with three partly conflicting ecosystem services (wood production, moose hunting and
344 livestock grazing), and its effect on total net present value. ‘Baseline’ resembles contemporary settings, while ‘lower’ and ‘upper’ are (realistic) extremes. The objective was to
345 maximize total net present value throughout a planning period (30 years, 3% interest rate), with and without minimum multiuse conditions (TRI-L = at least 50 moose hunted¹, 100
346 cattle and 1 000 sheep pastured each year; TRI-H = 150 moose, 300 cattle and 3 000 sheep). By comparing the different scenarios, we can deduct the opportunity costs of taking
347 multiuse concerns into account. Illustrated for property size 67 000 ha (43 000 ha productive forest land).

Parameters	Baseline	Lower	€/ha (W, H, G %)	Upper	€/ha (W, H, G %)
Maximizing total net present value <i>without</i> imposing multiuse conditions (the TRI-0 scenario)					
Tree species distribution (spruce, pine, birch) (%) ²	60, 30, 10	10, 30, 60	4 411 (97.2, 0.9, 1.9)	30, 60, 10	6 994 (98.6, 0.5, 0.9)
Meat prices (moose, cattle, sheep) (€/kg)	12, 6, 4	3, 1.5, 1	5 838 (99.4, 0.2, 0.4)	60, 30, 20	6 385 (90.6, 3.2, 6.2)
Timber market value (€/m ³) ³	38	10	2 473 (96.7, 1.6, 1.7)	100	15 028 (99.2, 0.2, 0.6)
Damage intensity browsed pine (α in eq.5) ⁴	0.21	0.99	5 926 (98.0, 0.7, 1.3)	0.01	5 913 (98.0, 0.7, 1.3)
Spruce trampled/cattle-day ha ⁻¹ (θ in eq.6) (%) ⁵	0.6	0.1	5 929 (98.0, 0.7, 1.3)	3	5 878 (98.0, 0.7, 1.3)
Interest rate (% discounted per annum)	3	1	6 922 (98.0, 1.5, 0.5)	5	5 250 (98.0, 0.8, 1.2)
Planning period (years)	30	10	5 032 (98.7, 0.7, 0.6)	80	6 466 (97.5, 0.7, 1.8)
Total net present value (€/ha) (from wood W%, hunting H%, grazing G%)	5 923 (98.0, 0.7, 1.3)				
Maximizing total net present value given <i>low levels</i> of multiuse conditions (the TRI-L scenario)					
Tree species distribution (spruce, pine, birch) (%)	60, 30, 10	10, 30, 60	4 164 (97.7, 1.5, 0.8)	30, 60, 10	6 628 (98.7, 1.0, 0.3)
Meat prices (sheep, cattle, moose) (€/kg)	12, 6, 4	3, 1.5, 1	5 661 (99.6, 0.3, 0.1)	60, 30, 20	6 219 (88.6, 5.5, 5.8)
Timber market value (€/m ³)	38	10	2 444 (95.3, 3.2, 1.5)	100	14 508 (99.4, 0.5, 0.2)
Damage intensity browsed pine (α in eq.5)	0.21	0.99	5 730 (98.0, 1.1, 0.9)	0.01	5 653 (98.2, 1.1, 0.7)
Spruce trampled/cattle-day ha ⁻¹ (θ in eq.6) (%)	0.6	0.1	5 777 (98.0, 1.1, 0.9)	3	5 395 (97.9, 1.2, 0.9)
Total net present value (€/ha) (from wood W%, hunting H%, grazing G%)	5 711 (98.0, 1.1, 0.9)				
Maximizing total net present value given <i>higher levels</i> of multiuse conditions (the TRI-H scenario)					
Tree species distribution (spruce, pine, birch) (%)	60, 30, 10	10, 30, 60	3 339 (95.2, 3.6, 1.2)	30, 60, 10	5 557 (97.3, 2.0, 0.7)
Meat prices (sheep, cattle, moose) (€/kg)	12, 6, 4	3, 1.5, 1	5 125 (99.0, 0.8, 0.2)	60, 30, 20	5 831 (85.6, 11.0, 3.3)
Timber market value (€/m ³)	38	10	2 290 (93.7, 4.8, 1.6)	100	13 145 (98.8, 0.9, 0.3)
Damage intensity browsed pine (α in eq.6)	0.21	0.99	5 312 (97.0, 2.3, 0.7)	0.01	5 005 (96.9, 2.4, 0.8)
Spruce trampled/cattle-day ha ⁻¹ (θ in eq.6) (%)	0.6	0.1	5 405 (97.1, 2.2, 0.7)	3	4 393 (96.4, 2.7, 0.9)
Total net present value (€/ha) (from wood W%, hunting H%, grazing G%)	5 231 (97.0, 2.3, 0.8)				

¹ Given that moose fecundity stays ≥ 0.5 calves/cow, cow: bull ratio stays ≤ 1.8 and no calves are orphaned due to hunting

² Proportion of ‘vegetation type’ in forest classified by the dominant tree of commercial timber interest

³ Net income = revenue minus harvesting costs. Value shown is for prima quality pine, but is stratum-specific in the model

⁴ Number of browsed pines determined by moose density/carrying capacity. When α approaches 1, all browsed pines are damaged, i.e. lose all monetary value

⁵ Proportion of (new) trees in stages I and II that will be trampled (and lose all monetary value) per cattle-day (influenced by cattle density and carrying capacity in the model)

349 **Table 2.** Compromising between three partly conflicting ecosystem services in forests (wood production, moose
350 hunting and livestock grazing), by maximizing a relative index denoting the weighted sum of realized proportion
351 of potential performance of each service (equal or unequal weighting of services). Performance throughout a
352 planning period of 30 years. Percentages are realized proportions for specific services, e.g. F^*/F_{max} for wood,
353 where F_{max} is the potential as found by maximizing wood performance in a separate scenario, and F^* is the same
354 metric to be jointly maximized using $I = F^*/F_{max} + C^*/C_{max} + S^*/S_{max} + M^*/M_{max}$ (thus, a 0-1 scale, where 1 is max).
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Objective	Performance (I)	Total (I)	Wood €/ha ¹ (%)	Cattle kg/ha (%)	Sheep kg/ha (%)	Moose kg/ha (%)
Maximize total I (all $w_i=1$)	0.55	5115 (88%)	1.2 (12%)	17.6 (85%)	6.9 (36%)	
Maximize I, weight cattle ² $w_c=2$	0.6	4233 (73%)	9.6 (92%)	2.9 (14%)	5.7 (30%)	
Maximize I, weight sheep ² $w_s=2$	0.63	5406 (93%)	0.4 (4%)	20.1 (97%)	4.6 (24%)	
Maximize I, weight moose ² $w_m=2$	0.55	4421 (76%)	1.6 (15%)	5.0 (24%)	15.6 (80%)	
Maximize I, weight moose ² $w_m=4$	0.66	3891 (67%)	0.0 (0%)	0.2 (1%)	19.1 (99%)	
Maximize wood ³ F^*/F_{max} (all $w_i=1$)	0.34	5809 (100%)	0.0 (0%)	5.4 (25%)	1.8 (9%)	
Maximize cattle ³ C^*/C_{max} (all $w_i=1$)	0.35	1773 (31%)	10.5 (100%)	0.1 (0%)	1.8 (9%)	
Maximize sheep ³ S^*/S_{max} (all $w_i=1$)	0.42	3342 (58%)	0.0 (0%)	20.8 (100%)	1.9 (10%)	
Maximize moose ³ M^*/M_{max} (all $w_i=1$)	0.32	1674 (29%)	0.0 (0%)	0.1 (0%)	19.4 (100%)	

¹ Net present value, with interest rate 3% and including expectation value

² These weights were arbitrarily chosen to show how different weighting affects I (and %), and do not indicate any kind of threshold levels. Weights of services not specified in a given scenario were set to 1 (only one service weighted differently in each scenario)

³ These scenarios are included to show how full potential realization of one service affects the potential realization of other services.

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357 3.5 Evaluating the opportunity cost of multiuse using normalized performance indices and weighting

358 A less skewed pattern of performance loss also emerged when using the normalized indices of realized
359 potential (Table 2, column 'Maximize total I') compared to when using a monetary measure with no
360 multiuse conditions (net present value, Fig. 4). The realized potential of each service (i.e. performance
361 loss) obtained with the normalized index most closely resembled the TRI-H scenario. Assigning
362 unequal weights to the services strongly affected their performance loss, particularly for cattle and
363 moose. It is noteworthy that weighted scenarios produced higher total I (see discussion).

364 **4 Discussion**

365 The output from our forest case system differed extensively when we changed the ecosystem service
366 to be prioritized. Wood production unequivocally yielded a higher total net present value, but led to a
367 substantial reduction in the production of goods and services from hunting and grazing. However, for
368 a wide range of parameter settings the inclusion of multiuse conditions (set as minimum performances
369 of the less profitable services) had minor impact on the net present value. These findings confirm other
370 studies showing that for many ecosystem services, a relatively small sacrifice by one stakeholder
371 group may secure large benefits to other users of the forest (e.g., Başkent et al. 2011; Duncker et al.
372 2012; Kyllönen et al. 2006; Soltani et al. 2014).

373 Any deviation from the maximization of total net value are difficult to accept for neo-classical
374 economists, as it dismisses the Pareto optimum, which is a deeply ingrained economic paradigm.
375 Resource allocation according to Pareto (1906) implies that optimality occurs when we cannot further
376 improve the wellbeing of one stakeholder without making at least one other stakeholder worse off. In
377 our forest case system, the Pareto optimum is represented by the TRI-0 scenario, i.e. maximizing for
378 total net present value with no minimum multiuse conditions. Clearly, moose hunters and cattle
379 owners would not receive much wellbeing if forest management should adhere only to a non-
380 compensating Pareto principle (Fig. 2B-C) (White 2009).

381 As expected, when we used the compromise programming technique to optimise multi-criteria
382 management of our case system, the unequal weighting of services strongly affected the performance
383 (see also Zekri & Romero 1993). Our case shows that the outcome of a given weighting is not
384 straightforward to predict when density dependent interactions are involved. For example, sheep
385 prioritizing ($w_s = 2$) also gave higher realization of wood potential, because more sheep meant less
386 moose and cattle and therefore reduced damage costs. Likewise, low-level moose prioritizing ($w_m = 2$,
387 but not $w_m = 4$) benefitted cattle, most likely because it facilitated a higher increase in the carrying
388 capacity than the moose could fully consume given the set of other constraints. In a practical
389 application of this sort of resource management, decision-makers must therefore engage in detailed
390 discussions about which weights to be used. In the case of a large forest property, the owner may make
391 the final decision unilaterally according to law. If too little weight is given to less superior
392 stakeholders, the owner may, however, end up in conflict with the local community. To maintain their

393 social capital in the local community owners could probably benefit from compromising somewhat on
394 the net present value (Bowles & Gintis 2002).

395 Because wood had such a superior income potential, prioritizing a single ecosystem service in our
396 study led to drastically different production of goods and services from hunting and grazing. This
397 inequality is analogous to many rural economies around the world. Smaller, often subsistence-oriented
398 stakeholders fall short if shared resources are distributed by monetary power only (Milner-Gulland
399 2011). On the other hand, while our study illustrates the beneficial potential of multiuse conditions
400 when dealing with conflicting ecosystem services, we should not lose sight of the fact that some
401 ecosystem services are best managed by land sparing, rather than land sharing (Phalan et al. 2011;
402 Vincent & Binkley 1993). Our results (Tables 1 and 2) indicate that cattle grazing may be such a
403 service when practiced in boreal forests where it is likely to contribute only a small part of total value,
404 with substantial negative impact on other services. In such scenarios, cattle grazing is better
405 undertaken on separate land outside the forest.

406 A shortcoming of our long-term planning approach is its lack of equations for dynamic
407 stakeholder behaviour. In reality, stakeholders are continuously receiving and acting from a range of
408 economic, social and cultural incentives (Bunnefeld & Keane 2014; Fulton et al. 2011). For example,
409 in our case study system it is unlikely that moose hunters will have the same hunting preferences in 20
410 years as they do today. The Nordic wood market currently fluctuates (Alajoutsijärvi et al. 2005), and
411 past predictability of forest owner behaviours may be disrupted (Follo 2011). The more qualitative-
412 oriented approaches to optimization modelling of ecosystem services now regularly address complex
413 stakeholder behaviour, e.g., with socioecological systems theory (SES, reviewed by Cumming 2011)
414 and management strategy evaluation (MSE, reviewed by Bunnefeld et al. 2011). Unfortunately,
415 studies incorporating stakeholder behaviour in a quantitative framework are generally lagging behind
416 the more conceptual and qualitative approaches (Redpath et al. 2015). We anticipate that our capacity
417 to better integrate social behaviour with both economics and ecology will follow as the emerging
418 research focus on quantitative multi-criteria modelling of ecosystem services catches up.

419 Although we in this study advocate using a quantitative model to aid ecosystem service
420 assessment, we do not argue for the exclusive use of such models. Decision-making regarding the
421 sustainable use of ecosystem services must always be founded in a set of adaptive processes

422 complementing each other (Argent 2009), as there are shortcomings associated with any single model.
423 The scientific and social processes vital to adaptive management can be broadly summarized as: a)
424 Identifying the appropriate spatiotemporal scales of each management option, b) retaining a focus on
425 statistical power and controlled experiments when selecting input data, c) scenario modelling to
426 outline potential outcome of the various management options, d) using model output to synthesize
427 socioecological consensus on the most relevant options, e) evaluating strategic alternatives for
428 achieving these management options, and f) communicating alternatives to the political arena for
429 negotiation and ultimate selection. The link between stages c) and d) is particularly critical (Mapstone
430 et al. 2008), and largely denotes where science ends and politics begin. Without a certain level of
431 stakeholder consensus, the political decisions will be hampered, and if a decision is reached
432 nevertheless, it is bound to exacerbate rather than mitigate conflict (Redpath et al. 2015).

433

434 *Conclusions*

435 The results of our study illustrate how a relatively small effort by one party (forest owners in our
436 example) may secure large benefits to others (local hunters or livestock owners in our example). Our
437 model approach should have the potential to mitigate conflicts of interests by providing more
438 comprehensive metrics, thus feeding broader acceptance into the larger scheme of adaptive
439 management processes. Provided there is sufficient empirical embedment of parameters, particularly
440 the biological ones, trade-off models have indeed proven to be a useful way of mitigating conflicts
441 over ecosystem services proactively rather than by remediation (Reed 2008).

442

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449

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Table A.1 Initial (at first year of planning period) and fixed conditions in an optimization model for trading off three partly conflicting forest ecosystem services (Nordic boreal forest as a case system): wood production, game hunting and livestock grazing. Parameters that we investigated for a range of values (with sensitivity analyses) are not listed here, but given in A.2.

Forest parameters	Value
Initial forest stage distribution (I, II, III, IV) ¹ (%) - <i>to be optimized by model</i>	5, 25, 55, 15
Tree density of forest stages (I, II, III, IV) ² (per ha) (td_s in eq. A9) - <i>fixed</i>	1 167, 983, 739, 372
Annual self-thinning of forest stages (I, II, III, IV) ² (%) - <i>fixed</i>	13.33, 3.94, 0.42, 0.39
Timber volume/tree in stage III (spruce, pine, birch) ² (m ³) - <i>fixed</i>	0.09, 0.16, 0.13
Timber volume/tree in stage IV (spruce, pine, birch) ² (m ³) - <i>fixed</i>	0.29, 0.38, 0.22
Recruitment cost (€/plant) (spruce) (cr eq. 4) - <i>fixed</i>	0.25
Fixed harvesting cost of logging in a stratum per annum (€) (cf in eq. 4) - <i>fixed</i>	950
Fixed annual administrative cost (€) (af in eq. 4) - <i>fixed</i>	100 000
Moose parameters	
Initial density (animals·km ⁻²) - <i>to be optimized by model</i>	0.9
Initial body mass (calf, yearling, cow, bull) (meat kg) - <i>to be optimized by model</i>	70, 150, 195, 235
Density effect on moose body mass (calf, yearling, cow, bull) (η_k in eqs. 1, A11) - <i>fixed</i>	4.9, 5.6, 8.8, 9.6
Primary fecundity (calves/cows before density effects) (f in eq. A2) - <i>fixed</i>	1.0
Annual survival without hunting (calves, older) (%) (o in eq. A2) - <i>fixed</i>	90, 97
Thinning factor in moose damage on pine (ψ_s in eq. 5) - <i>fixed</i>	0.52
Relative damage intensity (calf, yearling, adult) (b in eq. 5) - <i>fixed</i>	0.5, 1.0, 1.5
Niche overlap moose to sheep and cattle (see eq. A8) ³ - <i>fixed</i>	0.2, 0.05
Livestock parameters	
Number of days in forest pasturing season ($pdays$ in eqs. 2, 3, 6) - <i>fixed</i>	100
Initial cattle density (animals·km ⁻²) - <i>to be optimized by model</i>	1.0
Initial suckling body mass (meat kg) - <i>to be optimized by model</i>	120
Density effect on cattle body mass (calf, cow) (η_k in eq. 2) - <i>fixed</i>	3.3, 5.9
Primary fecundity cattle (calves/cow) (f in eq. A4) - <i>fixed</i>	1.0
Annual cattle survival without slaughter (calves, older) (%) (o in eq. A4) - <i>fixed</i>	90, 95
Initial sheep density (animals·km ⁻²) - <i>to be optimized by model</i>	9.7
Initial lamb body mass (carcass kg) - <i>to be optimized by model</i>	20
Density effect on sheep body mass (lamb, ewe) (η_k in eq. 3) - <i>fixed</i>	0.4, 0.7
Primary fecundity sheep (lambs/ewe) (f in eq. A4) - <i>fixed</i>	1.6
Annual sheep survival without slaughter (lambs, ewes) (%) (o in eq. A4) - <i>fixed</i>	80, 90
Niche overlap cattle to sheep, moose, or sheep to cattle, moose (see eq. A8) ⁴ - <i>fixed</i>	8, 0.67, 0.125, 0.083
Thinning factor in cattle damage on spruce (ψ_s in eq. 6) - <i>fixed</i>	0.46

¹ Approximates age span (varies slightly with Site Index and tree species): I = 0-3 years, II = 4-15 years, III = 16-60, IV = 61+

² These are stratum-specific (varies with Site Index and tree species), but for readability we show averages across the strata

³ Convert one moose into one sheep (or cattle) equivalent based on body mass difference and degree of niche overlap.

⁴ Convert one cattle (or sheep) into one moose equivalent based on body mass difference and degree of niche overlap.

Table A.2 Varying parameter values in an optimization model for trading off three partly conflicting forest ecosystem services (Nordic boreal forest as a case system): wood production, game hunting and livestock grazing, and its effect on total net present value. ‘Baseline’ resembles contemporary settings, while ‘lower’ and ‘upper’ are realistic extremes. The objective was to maximize net present value of each given service (see Table 1 for the outcome of optimizing from a combined perspective).

Maximizing wood (the WOOD scenario)	<i>Baseline</i>	<i>Lower</i> €/ha (W%)	<i>Upper</i> €/ha (W%)
Timber market value (€/m ³) ¹	38	10 2 450 (97.7)	100 14 980 (99.5)
Site Index (low, intermediate high) (%) ²	20,70,10	70,20,10 4 438 (98.4)	10,20,70 7 733 (99.2)
Tree species (pine, spruce, birch) (%) ³	60,30,10	10,30,60 4 364 (98.4)	30,60,10 6 955 (99.1)
Damage intensity browsed pine (α in eq. 5) ⁴	0.21	0.99 5 883 (98.8)	0.01 5 769 (98.8)
Spruce trampled/cattle-day ha ⁻¹ (θ in eq. 6) ⁵	0.6	0.1 5 878 (99.0)	3 5 834 (98.8)
Total net present value over 30 years (€/ha)	5 870		
% of total value stemming from wood (W%) ⁶	98.8		
Maximizing hunting (the HUNT scenario)⁷	<i>Baseline</i>	<i>Lower</i> €/ha (H%)	<i>Upper</i> €/ha (H%)
Per animal hunting fee bull, others (€)	456, 152	100, 30 2 167 (11.3)	2 000, 700 2 136 (18.6)
Moose meat price (€/kg)	12	3 1 841 (5.6)	60 3 169 (37.3)
Basic carrying capacity Kb_m (moose·km ⁻²)	2	1 2 049 (11.2)	4 2 261 (15.5)
Added carrying capacity ε_s (moose·km ⁻²) ⁸	4	2 2 005 (10.8)	8 2 343 (17.0)
Proportion of pine in forest (%) ⁹	30	10 2 243 (11.6)	90 2 211 (15.2)
Site Index (low, intermediate, high) (%)	20,70,10	70,20,10 1 170 (24.1)	10,20,70 3 035 (8.8)
Damage intensity browsed pine (α in eq. 5)	0.21	0.99 2 802 (9.7)	0.01 1 615 (16.8)
Total net present value over 30 years (€/ha)	2 148		
% of total value stemming from hunting (H%)	12.7		
Maximizing grazing (the GRAZ scenario)	<i>Baseline</i>	<i>Lower</i> €/ha (G%)	<i>Upper</i> €/ha (G%)
Meat prices (cattle, sheep) (€/kg)	6, 4	1.5, 1 3 345 (0.7)	30, 20 3 759 (11.6)
Basic carrying capacity Kb_c, Kb_s (stock·km ⁻²)	0.5, 5	0.2, 2 3 436 (2.4)	1.5, 15 3 571 (2.8)
Added carrying capacity ε_s (sheep, cattle·km ⁻²) ⁸	5, 50	1.7, 17 3 558 (1.1)	15, 150 3 668 (3.6)
Proportion of spruce in forest (%)	60	10 3 114 (2.7)	90 3 286 (3.0)
Site Index (low, intermediate, high) (%)	20,70,10	70,20,10 2 615 (3.6)	10,20,70 4 821 (1.7)
Spruce trampled/cattle-day ha ⁻¹ (θ in eq. 6) (%)	0.6	0.1 3 544 (2.5)	3 2 766 (3.2)
Total net present value over 30 years (€/ha)	3 410		
% of total value stemming from grazing (G%)	2.6		

¹ Net income = revenue minus harvesting costs. Value shown only for prima quality pine (stratum-specific in the model)

² The site’s inherent potential to produce timber (low = H40 7-11, intermediate = H40 14-17, high = H40 21, Tveite 1977)

³ Proportion of ‘vegetation type’ in forest classified by the dominating tree of commercial timber interest

⁴ Number of browsed pines determined by the ratio of moose density relative to its carrying capacity. When α approaches 0, all browsed pines are damaged, i.e. lose all monetary value

⁵ Proportion of (new) trees in stages I-II trampled per cattle-day (also influenced by cattle density and carrying capacity)

⁶ The remainder stemming from moose and/or livestock grazing. Interest rate set to 3% per annum. Net present values are per ha productive forest (calculated for a model property of 43 000 ha)

⁷ Given that moose fecundity stays ≥ 0.5 calves/cow, cow: bull ratio stays ≤ 1.8 and no calves are orphaned due to hunting

⁸ Stratum-specific number of animals/km² added per ha forest in stage $k = I, II, IV$ (averages shown across strata and stages)

⁹ Other tree species of commercial timber interest are spruce and birch (50: 50 of remaining proportion)

Specification of model algorithms

$M_{t,k}$ is the number of moose in the forest at time t , consisting of $k =$ five stages: 1 = calves (0-1 years old), 2 and 3 = female, respectively male yearlings (1-2 years old), 4 and 5 = cows, respectively bulls (2+ years old). If $M_t = \sum_{k=1}^K M_{t,k}$ is the total number of moose, then:

$$\mathbf{M}_{t+1}^{\mathbf{P}} = \mathbf{M} \cdot \mathbf{M}_t^{\mathbf{P}} - \mathbf{H}_t^{\mathbf{P}} \quad (\text{A1})$$

where $\mathbf{M}_t^{\mathbf{P}}$ is the vector of population stage structure at time t , $\mathbf{H}_t^{\mathbf{P}}$ is hunting stage structure and \mathbf{M} is the population projection matrix:

$$\mathbf{M} = \begin{bmatrix} 0 & 0 & 0 & f & 0 \\ \frac{o_1}{2} & 0 & 0 & 0 & 0 \\ \frac{o_1}{2} & 0 & 0 & 0 & 0 \\ 0 & o_2 & 0 & o_4 & 0 \\ 0 & 0 & o_3 & 0 & o_5 \end{bmatrix}, o_k \text{ and } f \geq 0 \quad (\text{A2})$$

where f is calves produced per cow (2+ years) per year (primary fecundity, see later for density effects), and o_k is the probability for moose in stage k to survive until next year given they are not hunted. We assume that calves contribute 50: 50 to the male and female yearling segments.

We project the livestock population in the same way, with sheep and cattle kept in separate matrices:

$$\mathbf{S}_{t+1}^{\mathbf{W}} = \mathbf{S} \cdot \mathbf{S}_t^{\mathbf{W}} - \mathbf{A}_t^{\mathbf{W}}, \quad \mathbf{C}_{t+1}^{\mathbf{W}} = \mathbf{C} \cdot \mathbf{C}_t^{\mathbf{W}} - \mathbf{B}_t^{\mathbf{W}} \quad (\text{A3})$$

where $\mathbf{S}_t^{\mathbf{W}}$ and $\mathbf{C}_t^{\mathbf{W}}$ are vectors of the population stage structure at time t , $\mathbf{A}_t^{\mathbf{W}}$ and $\mathbf{B}_t^{\mathbf{W}}$ are slaughter stage structures and \mathbf{S} and \mathbf{C} are projection matrices, with new animals recruited from the existing stock:

$$\mathbf{S} = \begin{bmatrix} 0 & 0 & \frac{f}{2} \\ 0 & 0 & \frac{f}{2} \\ o_1 & 0 & o_3 \end{bmatrix} \text{ (sheep), } o_k \text{ and } f \geq 0 \quad \mathbf{C} = \begin{bmatrix} 0 & 0 & 0 & \frac{f}{2} \\ 0 & 0 & 0 & \frac{f}{2} \\ o_1 & 0 & 0 & 0 \\ 0 & 0 & o_3 & o_4 \end{bmatrix} \text{ (cattle), } o_k \text{ and } f \geq 0 \quad (\text{A4})$$

where f is the number of offspring let out on pasture per female each year (determined by *ex situ* husbandry practices, thus treated as constants in the model), and o_k is the probability for animals in stage k to survive until next year given they are not slaughtered. The cattle population C_t has four stages: 1 = female sucklings (0-1 years old), 2 = male sucklings (0-1 years old), 3 = female heifers (1-2 years old), and 4 = cows (2+ year old females). Only cows 2+ years may reproduce. The sheep population S_t has the same stages, but 3 and 4 are grouped because sheep give birth as yearlings.

$F_{t,s}$ is the number of trees in forest stratum s at time t . Then:

$$\mathbf{F}_{t+1,s}^p = \mathbf{F} \cdot \mathbf{F}_{t,s}^p - \mathbf{U}_{t,s}^p + \mathbf{R}_{t+1,s}^w \quad (\text{A5})$$

where $\mathbf{F}_{t,s}^p$ is the vector of stratum stage structure at time t , $\mathbf{U}_{t,s}^p$ is harvesting stage structure and $\mathbf{R}_{t+1,s}^w$ is recruitment stage structure. The stratum projection matrix \mathbf{F} is given as:

$$\mathbf{F} = \begin{bmatrix} (1-g_1)o_1 & 0 & 0 & 0 \\ g_1o_1 & (1-g_2)o_2 & 0 & 0 \\ 0 & g_2o_2 & (1-g_3)o_3 & 0 \\ 0 & 0 & g_3o_3 & g_4o_4 \end{bmatrix} \quad (\text{A6})$$

where o_i is the probability for trees in stratum stage i to survive until next year given they are not harvested and g_i is their probability to grow into the next stage. Note that the forest growth is stationary. A stratum follows a pre-scheduled growth development through the stages, where natural and selective thinning are included in the mortality factor. This is a reasonable simplification, as we do not focus on forest yield *per se* here. Tree density and harvestable volume per tree vary between, but not within stages (Table A.1, volume is shown only for stages III-IV, as I-II have no market value).

We include density dependent interactions to the population projections using aggregated functions, i.e. the weighted sum of all individuals across all stages. In the following, we illustrate these equations for moose only, because the same principle underlies all density dependencies in the model:

$$\mathbf{M}_{t+1}^p = \mathbf{M}_t^p + D(M_t) \cdot (\mathbf{M} - \mathbf{I}) \cdot \mathbf{M}_t^p - \mathbf{H}_t^p \quad (\text{A7})$$

where \mathbf{I} is the identity matrix of \mathbf{M} (thus, $\mathbf{M}-\mathbf{I}$ parallels the intrinsic rate of population increase in a non-limited habitat), and $D(M_t)$ is the density dependent function to be included. In the example given in eq. A7, $D(M_t)$ pertains to recruitment, and takes the form:

$$D(M_t) = 1 - (M_t + cm \cdot C_t + sm \cdot S_t) / Km_t, \quad M_t + cm \cdot C_t + sm \cdot S_t \leq Km_t \quad (\text{A8})$$

where cm and sm transform cattle and sheep individuals into moose equivalents. These interspecific constants are based on the species' differences in body mass, and also their degree of niche overlap. For clarity, we denote animals simply by M , C and S in the remaining equations, but the model was run throughout with the adjusted sum of animals as outlined in eq. A8. Similar density functions based on logistic growth operate on body mass in eq. 1-3 and forest damage in eq. 5-6, see main manuscript). The Km_t is the forest's capacity to sustain foraging ungulates (denoted Km , Ks and Kc for moose, sheep and cattle respectively) and consists of a) a basic carrying capacity (Kb_m), defined as the number of animals sustained when the entire forest is in the least forage producing stage (stage III), and b) added carrying capacity from forest stages other than stage III:

$$Km_t = \sum_{s=1}^S [Kb_m + \varepsilon_s \cdot F_{t,s} / td_s], s \in \{k = I, II, IV\} \quad (A9)$$

where ε_s adjusts the foraging value of a forest stage relative to stage III, in terms of animals sustained per ha ($F_{t,s}/td_s$ gives the area of forest in stage s at time t). Note that ε_s varies with both tree stratum and animal species.

We included expectation values in our model, i.e. the net present value of all future use of a given resource (hunting, livestock grazing or logging) in the forest (MEV , CEV , SEV and FEV in eq. 2-5 and in main manuscript). We included these in order to avoid a decimation of the resource at the end of the planning period. We calculated the forest expectation value based on Svendsrud (2001) ⁱ:

$$FEV = \delta^T \cdot \sum_{s=1}^S \sum_{n=1}^N \left[\delta^{n-T} \cdot \left(H_{n,s} + \frac{H_{n,s}}{\delta^{-n} - 1} \right) - cr_s \cdot \frac{\delta^{-T}}{\delta^{-n} - 1} \right] \quad (A10)$$

where $H_{n,s}$ is the profit from harvesting a stratum at age n . We calculated this profit from the stratum-specific timber revenues (pf_s in eq. 5 in main manuscript) and volumes as given in Table A.1.

Expectation value thus increases with stage, promoting an older forest stage structure at the end of the planning period (depending on the rate of interest). In our model scenarios, the expectation value typically comprised approximately 10% of the total net present value stemming from logging. The cost of recruiting new forest cr_s (only for spruce, eq. 4) is calculated per strata by scaling the cost of buying nursery-grown spruce saplings (€/plant) (cr in Table A.1) by strata-specific tree densities.

There is no tradition for calculating expectation values of animals, so we apply a parallel to the calculations established in forestry, with the assumption that all future carrying capacities and hunting or slaughter stage structures are stationary on annual basis (calculation illustrated for moose only):

$$MEV = \delta^T \cdot \left[\sum_{k=1}^K [ph_k + pm \cdot wm_k \cdot [1 + \eta_k \cdot (M_t / Km_t)^{\rho_k}]^{-1}] / r \right] \quad (A11)$$

where r is the rate of interest. The higher the rate, the less influence MEV will have on the moose density at the end of the planning period. Because the value of a moose in a given life stage is the same throughout time for a constant Km_t , expectation values can be calculated only for the time T (= last year of planning period), and do not need to be summed over values calculated for each year.

ⁱ Svendsrud A (2001) Tabeller for beregning av verdien av skogbestand. Rapport fra skogforskningen. Suppl. 17.