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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. Presenting a socio-ecologically integrated trade-off model, and using the boreal
14 forest as a case, we outline the performance of partly competing services (game hunting, livestock
15 grazing and wood) when land sharing is the preferred option. Drawing on multi-criteria analyses
16 (MCA), we made factorial comparisons of both monetary (net present value) and non-monetary (e.g.,
17 number of game, livestock meat) output from scenarios with contrasting service priorities. Wood
18 production unequivocally yielded the highest net present value, but led to a substantial reduction in the
19 performance of hunting and grazing. By imposing multiuse conditions set as minimum performance of
20 the less profitable services, we evaluated the opportunity costs of multiuse without direct pricing of
21 non-commodities. We also quantified normalized indices of realized performance potential to evaluate
22 the cost of multiuse with a single, joint metric. Both approaches clearly and consistently show how the
23 forest owner's accepting a relatively small loss in one service may secure large gains in other services.
24 By democratically providing a comprehensive monetary and non-monetary evaluation, our approach
25 should generate broader acceptance for the decisional metrics among stakeholders. It thereby has the
26 potential to mitigate conflicts, feeding into the larger scheme of adaptive management.

27

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

29 1 Introduction

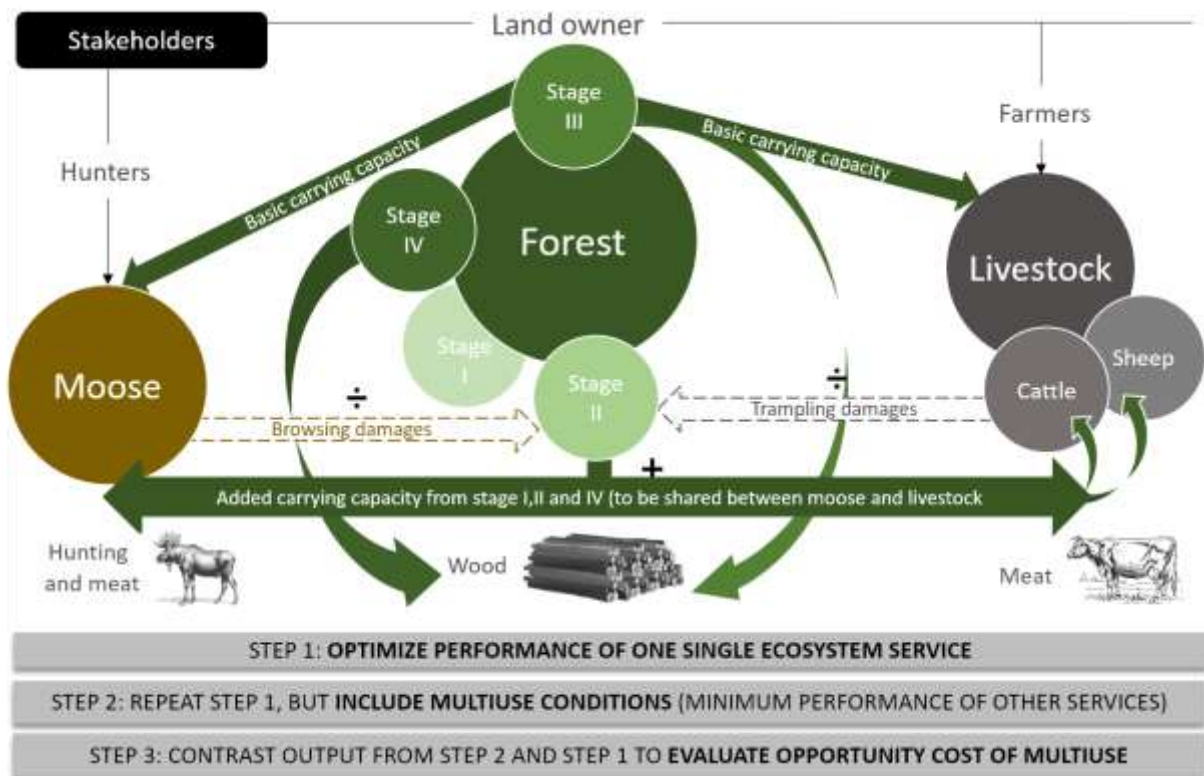
30 With a steadily rising human population and increasing needs for renewable resources, policymaking
31 for ecosystems services is more challenging than ever (Lindenmayer et al. 2012). Such intensification
32 of pressures on resources raises the potential for conflict between stakeholder interests,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

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

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

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

65 Aiming at socio-ecological integration, we outline a dynamic trade-off model for the optimization
66 of ecosystem services with partly conflicting stakeholder interests, when land sharing is the preferred
67 option. The inclusion of non-monetary goals and concerns adds new dimensions to the underlying
68 traditional Pareto optimization. Drawing on goal programming (Tamiz et al. 1998), we made factorial
69 comparisons of both monetary and non-monetary output from scenarios with contrasting service
70 priorities. By imposing multiuse conditions set as minimum performance of the less profitable
71 services, we evaluated the opportunity costs of multiuse without direct pricing of the non-commodities
72 (Fig. 1). Drawing also on elements from compromise programming (Zeleny 1974), we additionally
73 quantified normalized indices of realized performance potential to evaluate the cost of multiuse with a
74 single, joint measure. By democratically providing a comprehensive monetary and non-monetary
75 evaluation, our approach should generate broader stakeholder acceptance for the decisional metrics
76 (Ostrom 2007; Milner-Gulland 2011). It thereby has the potential to mitigate conflicts, feeding into the
77 larger schemes of adaptive management, such as the management strategy evaluation (Mapstone et al.
78 2008) or multi-criteria decision support (Kangas & Kangas 2005).



79

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

88 2 Model framework

89 2.1 Model objectives

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

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

108

109 2.2 Model structure

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

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

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

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

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

$$145 \quad \pi n = \sum_{t=1}^T \sum_{k=1}^K \delta^t \cdot \left[ph_k + pm \cdot wm_k \cdot \left[1 + \eta_k \cdot (M_t / Km_t)^{\rho_k} \right]^{-1} \right] \cdot h_{t,k} + MEV \quad (1)$$

$$146 \quad \pi c = \sum_{t=1}^T \sum_{k=1}^K \delta^t \cdot \left[pdays / 365 \cdot pc \cdot wc_k \cdot \left[1 + \eta_k \cdot (C_t / Kc_t)^{\rho_k} \right]^{-1} \right] \cdot sc_{t,k} + CEV \quad (2)$$

$$147 \quad \pi s = \sum_{t=1}^T \sum_{k=1}^K \delta^t \cdot \left[pdays / 365 \cdot ps \cdot ws_k \cdot \left[1 + \eta_k \cdot (S_t / Ks_t)^{\rho_k} \right]^{-1} \right] \cdot ss_{t,k} + SEV \quad (3)$$

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

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

$$160 \quad \pi f = \sum_{t=1}^T \sum_{s=1}^S \delta^t \cdot (pf_s \cdot u_{t,s} - cf_s - af - cr_s - cM_{t,s} - cC_{t,s}) + FEV \quad (4)$$

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

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

$$179 \quad cM_{t,s} = \delta^{T_H} \cdot \bar{p}f \cdot \psi_s \cdot f_{t,s} \cdot \sum_{k=1}^K (M_{t,k} \cdot b_k) / Km_t \cdot (1 + \alpha^{\beta \cdot M_t \cdot Km_t^{-1}})^{-1}, \quad s \in \{pine, k = II\} \quad (5)$$

$$180 \quad cC_{t,s} = \delta^{T_H} \cdot \bar{p}f \cdot \psi_s \cdot f_{t,s} \cdot \theta \cdot pdays \cdot C_t \cdot (f_{t,s} / td_{t,s})^{-1}, \quad s \in \{spruce, birch, k = I, II\} \quad (6)$$

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

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

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

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

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

$$207 \quad I = (w_h \cdot H^* / H_{\max} + w_c \cdot C^* / C_{\max} + w_s \cdot S^* / S_{\max} + w_f \cdot F^* / F_{\max}) / \sum_{i=1}^k w_i \quad (7)$$

208 where H_{\max} , C_{\max} , S_{\max} and F_{\max} are the potentials as found by maximizing each performance in
 209 individual model runs, H^* , C^* , S^* and F^* are the performances to be jointly maximized through the
 210 use of I , and w_i are weighting factors to prioritize ecosystem service i in relation to the other services.
 211 Each of the performance fractions (e.g., H^*/H_{\max}) as well as the joint metric I becomes a relative scale
 212 0-1, where 1 = maximum potential realized.

213

214 2.3 Model constraints set by non-commodity concerns

215 Not all elements of the forest ecosystem can be adequately addressed with economic theory (Wam
 216 2010). We set the following non-commodity concerns as model constraints (their effect on economic
 217 and biological output is addressed in our previous work, Wam & Hofstad 2007).

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

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

228

229 *2.4 Model parameterization and parameter sensitivity*

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

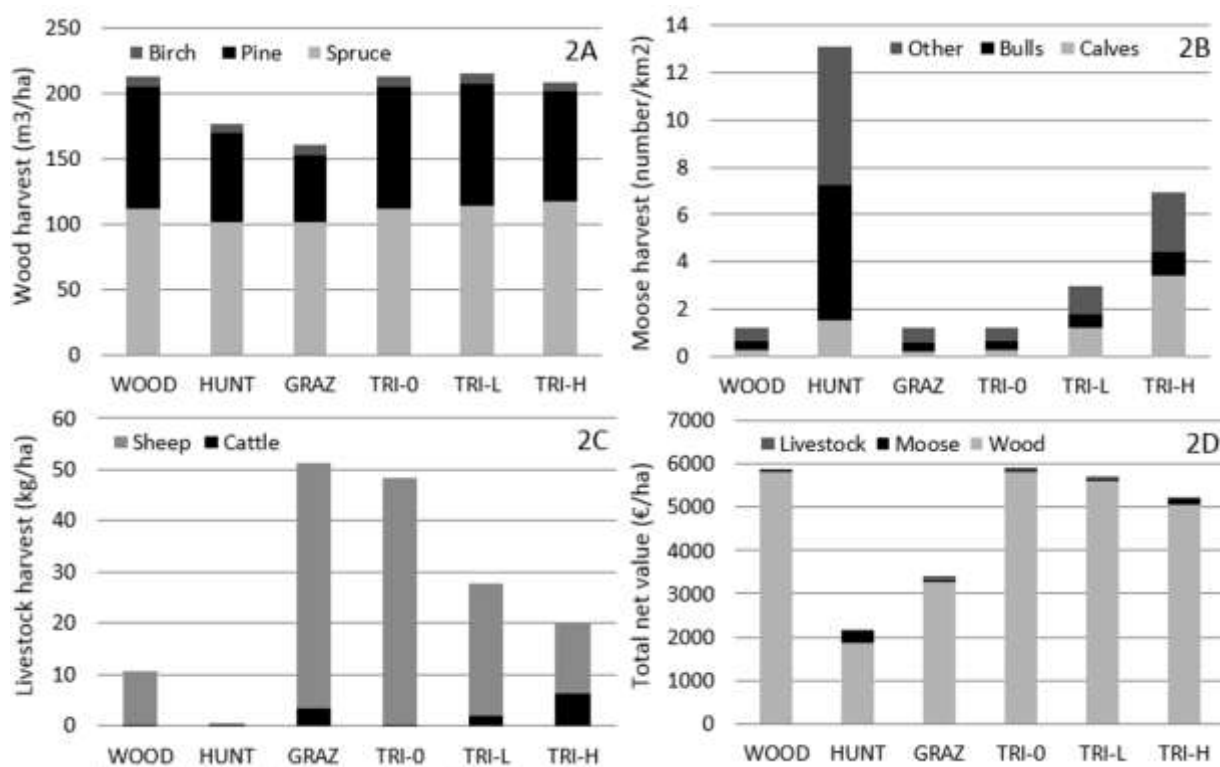
240 Forest growth, moose demography and in part moose: forest interactions were parameterized and
241 empirically validated in our earlier work (Wam & Hofstad 2007). The model was updated with new
242 field data on moose-forest interactions (Wam & Hjeljord 2010; Wam et al. 2010). We collected data
243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

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

254 **3 Results**255 **3.1 Prioritizing wood production (WOOD)**

256 Wood had about 2-3 times higher income potential than hunting and grazing (Fig. 2D), making it
 257 financially beneficial to minimize browsing and trampling damage. The optimal strategy both when
 258 maximizing net present value of wood (WOOD) and when maximizing total net present value (TRI-0),
 259 was therefore to eliminate moose and cattle, while keeping sheep at moderate densities (Fig. 2B-C). In
 260 the WOOD scenario, wood consistently contributed 98-99% of the total net present value over time,
 261 for the whole range of applied parameter settings (Table A.2). Factors facilitating contribution of
 262 wood to the total net present value (W%) were: a higher market value of timber, a higher Site Index
 263 (i.e. more productive forest land), and more pine in the forest. With all these facilitating factors
 264 combined, the WOOD scenario could generate a mean annual net value from wood production of 885
 265 €/ha (compared to 215 €/ha with parameters set at baseline).

266



267

268 **Fig. 2.** Potential performance (A-C) and total net present value (D) of forest ecosystem services over 30 years
 269 according to a socio-ecologically integrated trade-off model for partly conflicting services, with the objective to
 270 maximize net present value from wood production (WOOD), game hunting (HUNT), livestock grazing (GRAZ),
 271 or total net present value given various levels of multiuse conditions. TRI-0 = no such conditions; TRI-L = low
 272 levels (at least 50 moose hunted, 100 cattle and 1 000 sheep pastured each year; TRI-H = higher levels (at least
 273 150 moose, 300 cattle and 3 000 sheep). Illustrated for a land area of 67 000 ha (43 000 ha productive forest).

274 3.2 *Prioritizing game hunting (HUNT)*

275 The optimal strategy when prioritizing game hunting (HUNT) was to eliminate all livestock (Fig. 2C),
276 maintain spruce harvest and reduce pine harvest (Fig. 2A). Hunting contributed a highly variable share
277 of the total net present value, depending on parameter settings (Table A.2). Factors facilitating the
278 contribution of hunting (H%) to the total net present value were: a higher hunting revenue (more so for
279 fees paid per-kilo than per-capita), a higher carrying capacity, a lower Site Index, more pine in the
280 forest, and higher damage intensity on browsed pines. With all these facilitating factors combined, the
281 HUNT scenario could generate a mean annual net value from moose hunting of 100 €/ha (compared to
282 15 €/ha with parameters set at baseline), i.e. only a fraction of the potential from wood production.

283 While the wood harvest (m^3/ha) did not differ a lot between the HUNT and the WOOD scenarios,
284 the timber was logged at an earlier stage, facilitating shorter rotation times and larger areas being in
285 the more forage-productive younger stages. This and other (kb_m or ε_s , Table A.2) improvements of the
286 carrying capacity barely affected the total net present value, but greatly influenced the hunting
287 opportunities. The number of moose harvested in the HUNT scenario was ten times higher than in the
288 scenarios where moose was not explicitly prioritized (i.e. WOOD, TRI-0 and GRAZ) (Fig. 2B). Also,
289 a higher proportion of male moose (a target preferred by many hunters) was kept in the population as
290 well as harvested in the HUNT scenario compared to other scenarios.

291

292 3.3 *Prioritizing of livestock grazing (GRAZ)*

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

302 could generate a mean annual net value from sheep of 40 €/ha and 8 €/ha for cattle, compared to 4
303 €/ha and 3 €/ha with parameters at baseline (sheep and cattle prioritized in separate model runs).

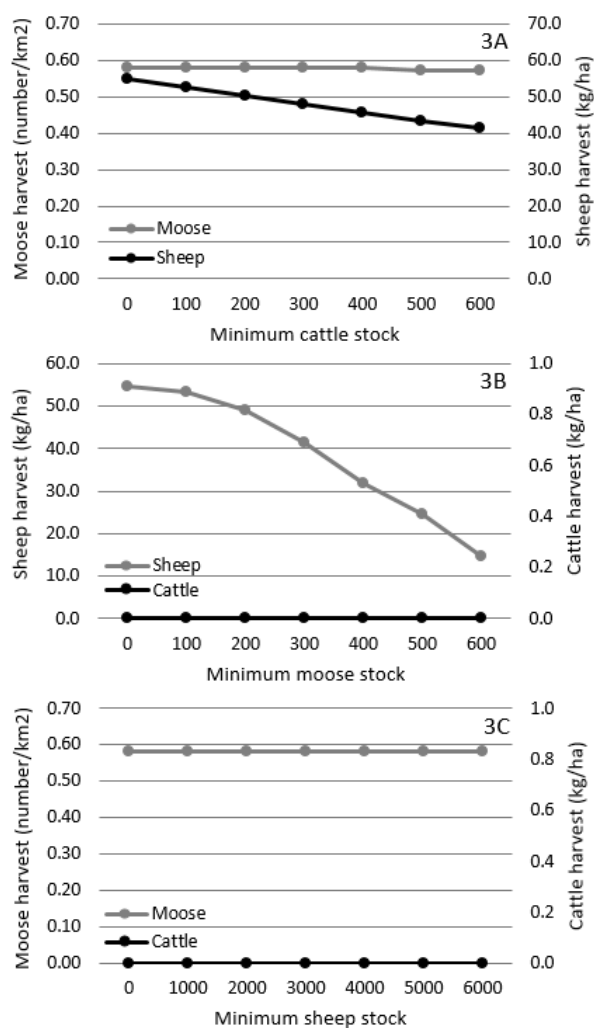
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305 3.4 Evaluating the opportunity cost of multiuse using minimum performance conditions (TRI-0, TRI-L, TRI-H)

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

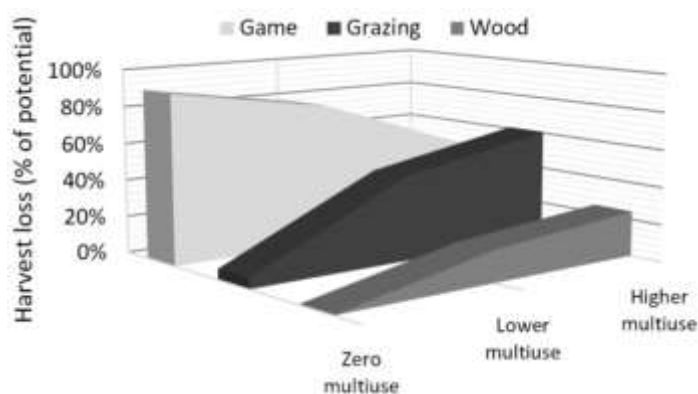
313 Compared to its effect on total net present value, adding multiuse conditions to the model more
314 strongly affected the biological output in terms of meat produced and game hunted. Raising the
315 minimum number of cattle in the forest had negligible influence on moose because of their low niche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

324 In contrast, raising the multiuse conditions to higher levels (TRI-H) generated a more fair
325 distribution of harvest loss (Fig. 4), still without jeopardizing much of the total net present value (see
326 Fig. 2D). Without multiuse conditions (TRI-0), game hunters carried practically all the burden of
327 being a less profitable stakeholder group. In TRI-0, their harvest was down by 90% compared to when
328 game hunting was prioritized. The wood production, on the other hand, was down by only about 20%
329 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).

345 **Table 1.** Varying parameter values in an optimization model for management of forests with three partly conflicting ecosystem services (wood production, moose hunting and
 346 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
 347 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
 348 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
 349 multiuse concerns into account. Illustrated for property size 67 000 ha (43 000 ha productive forest land).

Maximizing total net present value *without* imposing multiuse conditions (the TRI-0 scenario)

<i>Parameters</i>	<i>Baseline</i>	<i>Lower</i> €/ha (W, H, G %)	<i>Upper</i> €/ha (W, H, G %)
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.16) 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.17) (%) 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 *low levels* 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.16)	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.17) (%)	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 *higher levels* 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.16)	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.17) (%)	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)

351 **Table 2.** Compromising between three partly conflicting ecosystem services in forests (wood production, moose
 352 hunting and livestock grazing), by maximizing a relative index denoting the weighted sum of realized proportion
 353 of potential performance of each service (equal or unequal weighting of services). Performance throughout a
 354 planning period of 30 years. Percentages are realized proportions for specific services, e.g. F^*/F_{max} for wood,
 355 where F_{max} is the potential as found by maximizing wood performance in a separate scenario, and F^* is the same
 356 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)	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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359 3.5 Evaluating the opportunity cost of multiuse using normalized performance indices and weighting

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

366 4 Discussion

367 The output from our forest case system differed extensively when we changed the ecosystem service
368 to be prioritized. Wood production unequivocally yielded a higher total net present value, but led to a
369 substantial reduction in the production of goods and services from hunting and grazing. However, for
370 a wide range of parameter settings the inclusion of multiuse conditions (set as minimum performances
371 of the less profitable services) had minor impact on the net present value. These findings confirm other
372 studies showing that for many ecosystem services, a relatively small sacrifice by one stakeholder
373 group may secure large benefits to other users of the forest (e.g., Başkent et al. 2011; Duncker et al.
374 2012; Kyllönen et al. 2006; Soltani et al. 2014).

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

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

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

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

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

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

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

435

436 *Conclusions*

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

444

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451

452 **References**

- 453 Adamowicz W (2004) What's it worth? An examination of historical trends and future directions in
454 environmental valuation. *The Australian Journal of Agricultural and Resource Economics*, **48**,
455 419-444.
- 456 Alajoutsijärvi K, Holma H, Nyberg K, Tikkanen H (2005) Cyclicity in the Finnish and Swedish
457 sawmill industry, 1970–2000. *Scandinavian Economic History Review*, **53**, 66-90.
- 458 Arancibia F (2013) Challenging the bioeconomy: The dynamics of collective action in Argentina.
459 *Technology in Society*, **35**, 79-92.
- 460 Argent RM (2009) *Components of adaptive management*, in Adaptive Environmental Management.
461 Springer, Dordrecht.
- 462 Başkent EZ, Keleş S, Kadioğulları A, Bingöl Ö (2011) Quantifying the effects of forest management
463 strategies on the production of forest values: timber, carbon, oxygen, water, and soil.
464 *Environmental Modelling and Assessment*, **16**, 145-152.
- 465 Berge E (2002) Varieties of property rights to nature – some observations on landholding and resource
466 ownership in Norway and England, pages 14-33 in Schmithüsen F, Iselin G, Herbst P (eds) *Forest*
467 *Law and Environmental Legislation. Forstwissenschaftliche Beiträge 27*. Eidgenössische
468 Technische Hochschule Zürich.
- 469 Bioeconomy Council (2013) *Key issues paper of the bioeconomy council: "En route to the biobased*
470 *economy" (Areas of political and scientific priority 2013-2016)*. Bioeconomy Council, Berlin.
- 471 Bowles S, Gintis H (2002) Social capital and community governance. *The Economic Journal*, **112**,
472 F419-F436.
- 473 Brooke A, Kendrick D, Meeraus A, Raman R (1998) GAMS - user guide. GAMS Development
474 Corporation, Washington, USA.
- 475 Bunnefeld N, Hoshino E, Milner-Gulland EJ (2011) Management strategy evaluation: a powerful tool
476 for conservation? *TREE*, **26**, 441-447.
- 477 Bunnefeld N, Keane A (2014) Managing wildlife for ecological, socioeconomic, and evolutionary
478 sustainability. *PNAS*, **111**, 12964-12965.
- 479 Cumming GS (2011) *Spatial resilience in social-ecological systems*. Springer, London.

- 480 De Groot RS, Alkemade R, Braat L, Hein L, Willemsen L (2010) Challenges in integrating the concept
481 of ecosystem services and values in landscape planning, management and decision making.
482 *Ecological Complexity*, **7**, 260-272.
- 483 Dennis RA, Mayer J, Applegate G, Chokkalingam U, Colfer CP, Kurniawan I, Lachowski H, Maus P,
484 Permana RP, Ruchiat Y, Stolle F, Suyanto, Tomich TP (2005) Fire, people and pixels: linking
485 social science and remote sensing to understand underlying causes and impacts of fires in
486 Indonesia. *Human Ecology*, **33**, 465-504.
- 487 Drud A (2006) *Conopt*. ARKI Consulting and Development, Bagsvaerd.
- 488 Duncker PS, Raulund-Rasmussen K, Gundersen P, Katzensteiner K, De Jong J, Ravn HP, Smith M,
489 Eckmüllner O, Spiecker H (2012) How forest management affects ecosystem services, including
490 timber production and economic return: synergies and trade-offs. *Ecology and Society*, **17**, 50.
- 491 Fischer J, Manning AD, Steffen W, Rose DB, Danell K, Felton A, Garnett S, Gilna B, Heinsohn R,
492 Lindenmayer DB, MacDonald B, Mills F, Newell B, Reid J, Robin L, Sherren K, Wade A (2007)
493 Mind the sustainability gap. *TREE*, **22**, 621-624.
- 494 Follo G (2011) Factors influencing Norwegian small-scale private forest owners' ability to meet the
495 political goals. *Scandinavian Journal of Forest Research*, **26**, 385-393.
- 496 Fulton EA, Smith DM, Smith DC, van Putten IE (2011) Human behaviour: the key source of
497 uncertainty in fisheries management. *Fish and Fisheries*, **12**, 2-17.
- 498 Hajkowicz (2008) Rethinking the economist's evaluation toolkit in light of sustainability policy.
499 *Sustainability: science, practice & policy*, **4**, 17-24
- 500 Hardin G (1968) The tragedy of the commons. *Science*, **162**, 1243-1248.
- 501 Hjeljord O, Histøl T, Wam HK (2014) Forest pasturing of livestock in Norway: effects on spruce
502 regeneration. *Journal of Forestry Research*, **25**, 941-945.
- 503 Hotte L (2001) Conflicts over property rights and natural-resource exploitation at the frontier. *Journal*
504 *of Development Economics*, **66**, 1-21.
- 505 Jacobsen C (2014) Kommersialisering av jakt på elg og hjort : en studie av grunneiere og
506 jaktentreprenørers opplevelser av muligheter og utfordringer, med vekt på konflikter med andre
507 aktører [in Norwegian with English abstract]. Thesis, Norwegian University of Life Sciences, Ås.

- 508 Kangas A, Kangas J, Pykäläinen J (2001) Outranking methods as tools in strategic natural resource
509 planning. *Silva Fennica*, **35**, 215-227.
- 510 Kangas J, Kangas A (2005) Multiple criteria decision support in forest management - the approach,
511 methods applied, and experiences gained. *Forest Ecology and Management*, **207**, 133-143.
- 512 Kyllönen S, Colpaert A, Heikkinen H, Jokinen M, Kumpula J, Marttunen M, Muje K, Raitio K (2006)
513 Conflict management as a means to the sustainable use of natural resources. *Silva Fennica*, **40**,
514 687-728.
- 515 Leopold A (1949) A sand county almanac with essays on conservation from Round River. Reprint
516 1966. Oxford University Press, New York.
- 517 Leslie PH (1945) On the use of matrices in certain population mathematics. *Biometrika*, **3**, 183-212.
- 518 Lindenmayer D, Cunningham S, Young A (2012) *Land use intensification: Effects on agriculture,*
519 *biodiversity and ecological processes*. CSIRO Publishing, Melbourne.
- 520 MacKerron G (2012) Happiness economics from 35,000 feet. *Journal of Economic Surveys*, **26**, 705-
521 735.
- 522 Mapstone BD, Little LR, Punt AE, Davies CR, Smith ADM, Pantus F, McDonald AD, Williams AJ,
523 Jones A (2008) Management strategy evaluations for line fishing in the Great Barrier Reef:
524 Balancing conservation and multi-sector fishery objectives. *Fisheries Research*, **94**, 315-329.
- 525 Maxwell SL, Milner-Gulland EJ, Jones JPG, Knight AT, Bunnefeld N, Nuno A, Bal P, Earle S,
526 Watson JEM, Rhodes JR (2014) Being smart about SMART environmental targets *Science*, **347**,
527 1075-1076.
- 528 McKibben B (2007) *Deep economy: The wealth of communities and the durable future*. One World
529 Publications, Oxford.
- 530 Milner-Gulland EJ (2011) Integrating fisheries approaches and household utility models for improved
531 resource management. *Proceedings of the National Academy of Sciences*, **108**, 1741-1746.
- 532 OECD (2009) *The Bioeconomy to 2030: Designing a Policy Agenda*. OECD Publishing, Paris.
- 533 Ostrom E (2007) A diagnostic approach for going beyond panaceas. *Proceedings of the National*
534 *Academy of Sciences*, **104**, 15181-15187.
- 535 Pareto V (1906) *Manuale di economia politica*. Editrice Libreria, Milano.

- 536 Phalan B, Onial M, Balmford A, Green RE (2011) Reconciling food production and biodiversity
537 conservation: land sharing and land sparing compared. *Science*, **333**, 1289-1291.
- 538 Philibert C (2003) Discounting the future. International Society for Ecological Economics,
539 Encyclopaedia [Accessed October 14 2015 from <http://isecoeco.org/>]
- 540 Poff B, Teclé A, Neary DG, Gells B (2010) Compromise programming in forest management. *Journal*
541 *of Arizona-Nevada Academy of Science*, **42**, 44-60.
- 542 Redpath SM, Young J, Evely A, Adams WM, Sutherland WJ, Whitehouse A, Amar A, Lambert RA,
543 Linnell JDC, Watt A Gutiérrez RJ (2013) Understanding and managing conservation conflicts.
544 *Trends in Ecology & Evolution*, **28**, 100-109.
- 545 Redpath SM, Gutiérrez RJ, Wood KA, Young JC (eds) (2015) *Conflicts in conservation. Navigating*
546 *towards solutions*. Cambridge University Press, Cambridge.
- 547 Reed MS (2008) Stakeholder participation for environmental management: a literature review.
548 *Biological Conservation*, **141**, 2417-2431.
- 549 Rodríguez JP, Beard Jr TD, Bennett EM, Cumming GS, Cork SJ, Agard J, Dobson AP, Peterson GD
550 (2006) Trade-offs across space, time, and ecosystem services. *Ecology and Society*, **11**, 28.
- 551 Rodríguez JP, Beard Jr TD, Bennett EM, Cumming GS, Cork S, Agard J, Dobson AP, Schlueter M,
552 McAllister RRJ, Arlinghaus R, Bunnefeld N, Eisenack K, Hoelker F, Milner-Gulland EJ, Müller B,
553 Nicholson E, Quaas M, Stöven M (2012) New horizons for managing the environment: a review of
554 coupled social-ecological systems modelling. *Natural Resource Modelling*, **25**, 219-272.
- 555 Schlüter M, Hinkel J, Bots PWG, Arlinghaus R (2014) Application of the SES framework for model-
556 based analysis of the dynamics of social-ecological systems. *Ecology and Society*, **19**, 36.
- 557 Schuhmann PW, Mahon R (2015) The valuation of marine ecosystem goods and services in the
558 Caribbean: A literature review and framework for future valuation efforts. *Ecosystem Services*, **11**,
559 56-66.
- 560 Solberg EJ, Rolandsen CM, Heim M, Grøtan V, Garel M, Sæther B-E, Nilsen EB, Austrheim G,
561 Herfindal I (2006) Moose in Norway - an analysis of material collected by moose hunters 1966-
562 2004. *NINA Report*, **125**, 1-197.
- 563 Soltani A, Sankhayan PL, Hofstad O (2014) A dynamic bio-economic model for community
564 management of goat and oak forests in Zagros, Iran. *Ecological Economics*, **106**, 174-185.

- 565 Speed JD, Austrheim G, Hester AJ, Solberg EJ, Tremblay JP (2013) Regional-scale alteration of clear-
566 cut forest regeneration caused by moose browsing. *Forest Ecology and Management*, **289**, 289-
567 299.
- 568 Svendsrud A (2001) Tabeller for beregning av verdien av skogbestand. *Rapport fra skogforskningen*.
569 Suppl. 17. (in Norwegian only)
- 570 Sæther BE, Solberg EJ, Heim M (2003) Effects of altering sex ratio structure on the demography of an
571 isolated moose population. *Journal of Wildlife Management*, **67**, 455-466.
- 572 Sæther BE, Solberg EJ, Heim M, Stacy JE, Jakobsen KS, Olstad R (2004) Offspring sex ratio in
573 moose *Alces alces* in relation to paternal age: an experiment. *Wildlife Biology*, **10**, 51-57.
- 574 Tamiz M, Jones DF, Romero (1998) Goal programming for decision making: an overview of the
575 current state-of-the-art. *European Journal of Operational Research*, **111**, 569-581.
- 576 Tjosvold D (1991) Rights and responsibilities of dissent: cooperative conflict. *Employee*
577 *Responsibilities and Rights Journal*, **4**, 13-23.
- 578 Tveite B (1977) Site-index curves for Norway spruce (*Picea abies* (L.) Karst.) (in Norwegian with
579 English abstract). *Report Norwegian Forest Research Institute*, **33**, 1-84.
- 580 Usher MB (1966) A matrix approach to the management of renewable resources, with special
581 reference to selection forests. *Journal of Applied Ecology*, **3**, 355-367.
- 582 Usher MB (1969) A matrix approach to the management of renewable resources, with special
583 reference to selection forests – two extensions. *Journal of Applied Ecology*, **6**, 347-348.
- 584 Vincent JR, Binkley CS (1993) Efficient multiple-use forestry may require land-use
585 specialization. *Land Economics*, **69**, 370-376.
- 586 Wam HK, Hofstad O (2007) Taking timber browsing damage into account: a density dependent matrix
587 model for the optimal harvest of moose in Scandinavia. *Ecological Economics*, **62**, 45-55.
- 588 Wam HK (2010) Economists, time to team up with the ecologists! *Ecological Economics*, **69**, 675-
589 679.
- 590 Wam HK, Hjeljord O (2010) Moose summer and winter diets along a large scale gradient of forage
591 availability in southern Norway. *European Journal of Wildlife Research*, **56**, 745-755.
- 592 Wam HK, Hjeljord O, Solberg, EJ (2010) Differential forage use makes carrying capacity equivocal
593 on ranges of Scandinavian moose (*Alces alces*). *Canadian Journal of Zoology*, **88**, 1179-1191.

- 594 White M (2009) Numeraire illusion: The final demise of the Kaldor-Hicks principle. In *Theoretical*
595 *foundations of law and economics*. New York: Cambridge University Press.
- 596 Zekri S, Romero C (1993) Public and private compromises in agricultural water management. *Journal*
597 *of Environmental Management*, **37**, 281-290.
- 598 Zeleny M (1974) A concept of compromise solutions and the method of the displaced ideal.
599 *Computers and Operations Research*, **1**, 479-496.