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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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 ecosystems 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).
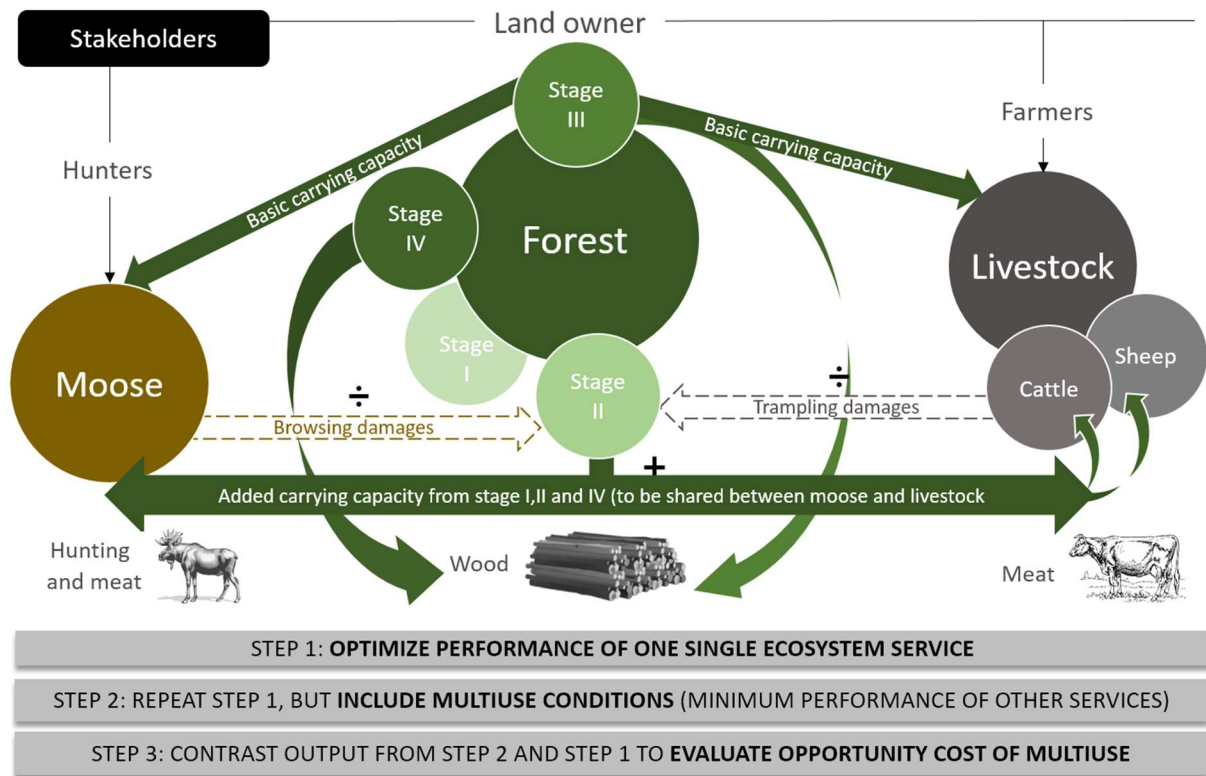
**Figure 1**. The use of one ecosystem service may both impede and facilitate other services, as partly illustrated above using forest as a case: wood logging in older forest (stage III-IV) substantially contributes to food carrying capacity for moose and livestock, but livestock cause trampling damages and moose cause browsing damage to the new recruitment of trees (stage I-II). In our trade-off model, we sequentially assess the effects of favouring single or all stakeholder groups on not only monetary output (net present value), but also goods and services (hunting, wood and meat). Because different stakeholder groups have different goals and gains, also of non-economic value, trading-off the 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 $Km$, $Ks$ and $Kc$ 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 $wm_k$, $wc_k$, $ws_k$). 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 (*πm, πc, πs*, respectively) (€) is:

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

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

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

146 where $\delta^t$ 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 $\eta_k$ and $\rho_k$ 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, $\eta_k$

152 and $\rho_k$ 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 ($\pi f$) is:

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

159 where $pf_s$ is the net revenue (harvesting costs deducted) (€) per m$^3$ of wood cut in stratum $s$, $u_{t,s}$ is the

160 amount of wood (m$^3$) 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    $$cM_{t,s} = \delta^{T_H} \cdot \overline{pf} \cdot \psi_s \cdot f_{t,s} \cdot \sum_{k=1}^{K} \left( M_{t,k} \cdot b_k \right) / Km_t \cdot (1 + \alpha^{\beta \cdot M_t \cdot Km_t^{-1}})^{-1} , \ s \in \left\{ pine, k = II \right\} \qquad (5)$$

178    $$cC_{t,s} = \delta^{T_H} \cdot \overline{pf} \cdot \psi_s \cdot f_{t,s} \cdot \theta \cdot pdays \, C_t \cdot (f_{t,s} / td_{t,s})^{-1} , \ s \in \left\{ spruce, birch, k = I, II \right\} \qquad (6)$$

179    where $\delta^{T_H}$ 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 $(\overline{pf})$ 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 $\alpha$ and $\beta$ 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, $\alpha$ and $\beta$ 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 $\psi_s$ (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 $\theta$ 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 $\psi_s$ (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    $$I = (w_h \cdot H^*/H_{\max} + w_c \cdot C^*/C_{\max} + w_s \cdot S^*/S_{\max} + w_f \cdot F^*/F_{\max}) / \sum_{i=1}^{k} w_i \qquad (7)$$

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

### 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.

## 3  Results

### 3.1  Prioritizing wood production (WOOD)

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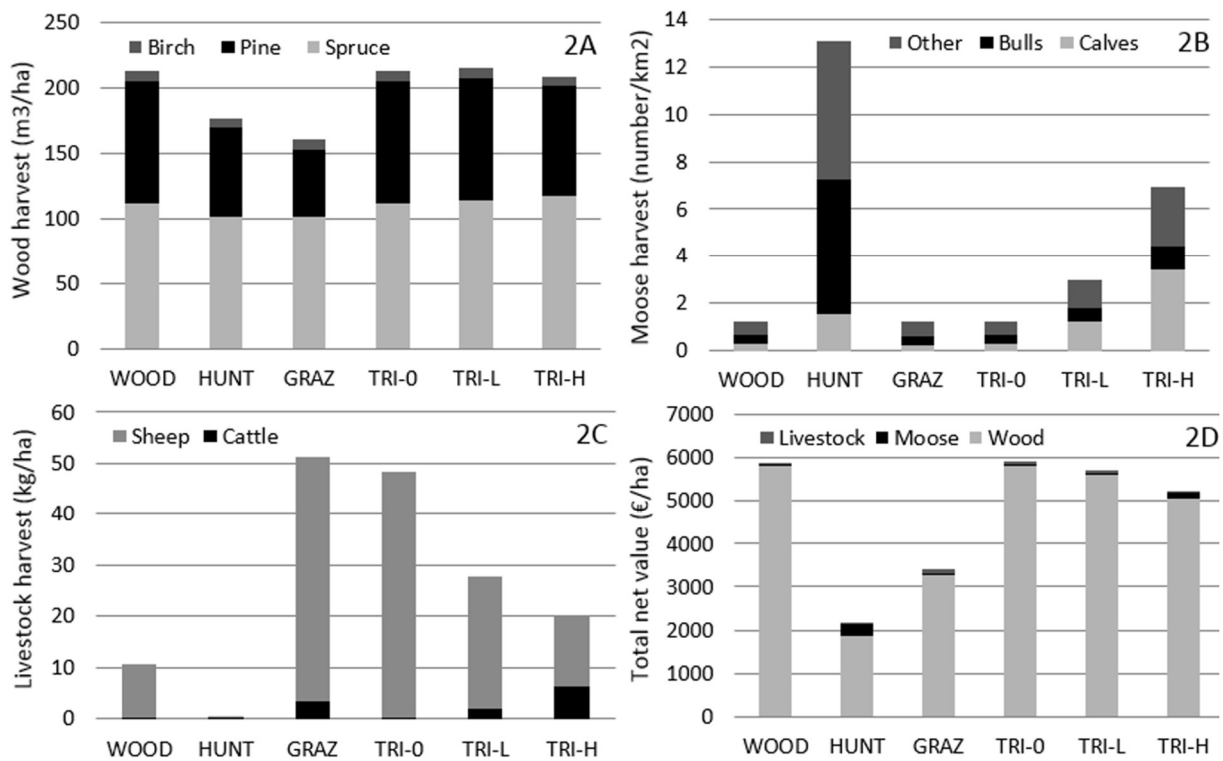


**Fig. 2**. Potential performance (A-C) and total net present value (D) of forest ecosystem services over 30 years according to a socio-ecologically integrated trade-off model for partly conflicting services, with the objective to maximize net present value from wood production (WOOD), game hunting (HUNT), livestock grazing (GRAZ), or total net present value given various levels of multiuse conditions. TRI-0 = no such conditions; TRI-L = low levels (at least 50 moose hunted, 100 cattle and 1 000 sheep pastured each year; TRI-H = higher levels (at least 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 ($m^3$/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 $\varepsilon_s$, 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).

302

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).
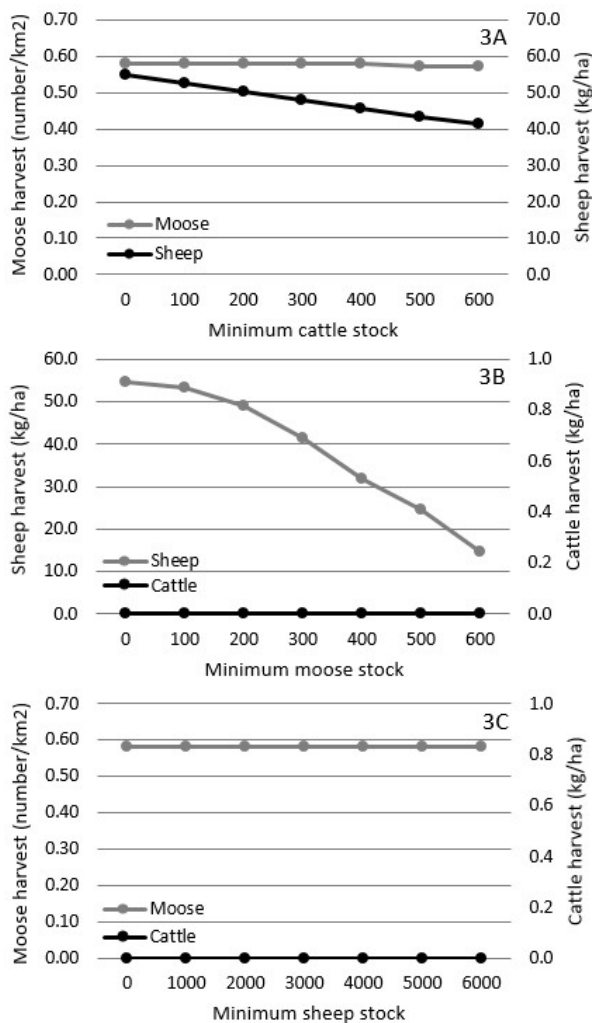
328
329
330  **Fig. 3**. Potential performance of forest ecosystem services over 30 years according to a socio-ecologically
331  integrated trade-off model for partly conflicting services (wood production, moose hunting and livestock
332  grazing), with the objective to maximize total net present value given various levels of multiuse conditions, i.e.
333  minimum performance of the monetarily less profitable services A) cattle, B) moose, and C) sheep (profit of
334  wood production was superior to that of moose and livestock, thus not favoured with multiuse conditions).



335

336  **Fig. 4**. Loss of potential performance from forest ecosystem services according to a socio-ecologically integrated
337  trade-off model for partly conflicting services (wood production, moose hunting and livestock grazing), with the
338  objective to maximize total net present value given three levels of multiuse conditions imposed to secure
339  minimum performance of the monetarily less profitable services (i.e. grazing and game). The harvest potential
340  (number of moose/km2, kg livestock meat/ha or m$^3$ of timber/ha) was calculated for a 30 year planning period,
341  and equals the performance obtained if the ecosystem service in question was completely prioritized (i.e.
342  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 hunted[1], 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 *without* 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 (€/m³)[3] | 38 | 10 | 2 473 (96.7, 1.6, 1.7) | 100 | 15 028 (99.2, 0.2, 0.6) |
| Damage intensity browsed pine ($\alpha$ 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 ($\theta$ 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 *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 (€/m³) | 38 | 10 | 2 444 (95.3, 3.2, 1.5) | 100 | 14 508 (99.4, 0.5, 0.2) |
| Damage intensity browsed pine ($\alpha$ 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 ($\theta$ 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 *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 (€/m³) | 38 | 10 | 2 290 (93.7, 4.8, 1.6) | 100 | 13 145 (98.8, 0.9, 0.3) |
| Damage intensity browsed pine ($\alpha$ 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 ($\theta$ 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)

348

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).
355

| Performance\Objective | 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.

356

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

357

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

443    **Acknowledgements**

449

450    **References**

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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] (%) - *to be optimized by model* | 5, 25, 55, 15 |
| Tree density of forest stages (I, II, III, IV)[2] (per ha) ($td_s$ in eq. A9) - *fixed* | 1 167, 983, 739, 372 |
| Annual self-thinning of forest stages (I, II, III, IV)[2] (%) - *fixed* | 13.33, 3.94, 0.42, 0.39 |
| Timber volume/tree in stage III (spruce, pine, birch)[2] (m³) - *fixed* | 0.09, 0.16, 0.13 |
| Timber volume/tree in stage IV (spruce, pine, birch)[2] (m³) - *fixed* | 0.29, 0.38, 0.22 |
| Recruitment cost (€/plant) (spruce) ($cr$ eq. 4) - *fixed* | 0.25 |
| Fixed harvesting cost of logging in a stratum per annum (€) *($cf$ in eq. 4) - fixed* | 950 |
| Fixed annual administrative cost (€) ($af$ in eq. 4) - *fixed* | 100 000 |

| Moose parameters | |
|---|---|
| Initial density (animals·km⁻²) - *to be optimized by model* | 0.9 |
| Initial body mass (calf, yearling, cow, bull) (meat kg) - *to be optimized by model* | 70, 150, 195, 235 |
| Density effect on moose body mass (calf, yearling, cow, bull) ($\eta_k$ in eqs. 1, A11) - *fixed* | 4.9, 5.6, 8.8, 9.6 |
| Primary fecundity (calves/cows before density effects) ($f$ in eq. A2) - *fixed* | 1.0 |
| Annual survival without hunting (calves, older) (%) ($o$ in eq. A2) - *fixed* | 90, 97 |
| Thinning factor in moose damage on pine ($\psi_s$ in eq. 5) - *fixed* | 0.52 |
| Relative damage intensity (calf, yearling, adult) ($b$ in eq. 5) - *fixed* | 0.5, 1.0, 1.5 |
| Niche overlap moose to sheep and cattle (see eq. A8)[3] - *fixed* | 0.2, 0.05 |

| Livestock parameters | |
|---|---|
| Number of days in forest pasturing season ($pdays$ in eqs. 2, 3, 6) - *fixed* | 100 |
| Initial cattle density (animals·km⁻²) - *to be optimized by model* | 1.0 |
| Initial suckling body mass (meat kg) - *to be optimized by model* | 120 |
| Density effect on cattle body mass (calf, cow) ($\eta_k$ in eq. 2) - *fixed* | 3.3, 5.9 |
| Primary fecundity cattle (calves/cow) ($f$ in eq. A4) - *fixed* | 1.0 |
| Annual cattle survival without slaughter (calves, older) (%) ($o$ in eq. A4) - *fixed* | 90, 95 |
| Initial sheep density (animals·km⁻²) - *to be optimized by model* | 9.7 |
| Initial lamb body mass (carcass kg) - *to be optimized by model* | 20 |
| Density effect on sheep body mass (lamb, ewe) ($\eta_k$ in eq. 3) - *fixed* | 0.4, 0.7 |
| Primary fecundity sheep (lambs/ewe) ($f$ in eq. A4) - *fixed* | 1.6 |
| Annual sheep survival without slaughter (lambs, ewes) (%) ($o$ in eq. A4) - *fixed* | 80, 90 |
| Niche overlap cattle to sheep, moose, or sheep to cattle, moose (see eq. A8)[4] - *fixed* | 8, 0.67, 0.125, 0.083 |
| Thinning factor in cattle damage on spruce ($\psi_s$ in eq. 6) – *fixed* | 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)** | *Baseline* | *Lower* | €/ha (W%) | *Upper* | €/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 ($\alpha$ in eq. 5)[4] | 0.21 | 0.99 | 5 883 (98.8) | 0.01 | 5 769 (98.8) |
| Spruce trampled/cattle-day ha$^{-1}$ ($\theta$ 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] | *Baseline* | *Lower* | €/ha (H%) | *Upper* | €/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 | 1841 (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 $\varepsilon_s$ (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 ($\alpha$ 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)** | *Baseline* | *Lower* | €/ha (G%) | *Upper* | €/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 $\varepsilon_s$ (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$^{-1}$ ($\theta$ 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/km$^2$ 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:

$$\overset{\upsilon}{\mathbf{M}}_{t+1} = \mathbf{M} \cdot \overset{\upsilon}{\mathbf{M}}_t - \overset{\upsilon}{\mathbf{H}}_t \qquad (A1)$$

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

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

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:

$$\overset{\omega}{\mathbf{S}}_{t+1} = \mathbf{S} \cdot \overset{\omega}{\mathbf{S}}_t - \overset{\omega}{\mathbf{A}}_t \; \text{(sheep)}, \quad \overset{\omega}{\mathbf{C}}_{t+1} = \mathbf{C} \cdot \overset{\omega}{\mathbf{C}}_t - \overset{\omega}{\mathbf{B}}_t \; \text{(cattle)} \qquad (A3)$$

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

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

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.

29

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

$$\vec{F}_{t+1,s} = \mathbf{F} \cdot \vec{F}_{t,s} - \vec{U}_{t,s} + \overset{\omega}{R}_{t+1,s} \tag{A5}$$

where $\vec{F}_{t,s}$ is the vector of stratum stage structure at time $t$, $\vec{U}_{t,s}$ is harvesting stage structure and

$\overset{\omega}{R}_{t+1,s}$ is recruitment stage structure. The stratum projection matrix $\mathbf{F}$ is given as:

$$\mathbf{F} = \begin{bmatrix} (1-g_1)o_1 & 0 & 0 & 0 \\ g_1 o_1 & (1-g_2)o_2 & 0 & 0 \\ 0 & g_2 o_2 & (1-g_3)o_3 & 0 \\ 0 & 0 & g_3 o_3 & g_4 o_4 \end{bmatrix} \tag{A6}$$

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:

$$\vec{M}_{t+1} = \vec{M}_t + D(M_t) \cdot (\mathbf{M} - \mathbf{I}) \cdot \vec{M}_t - \vec{H}_t \tag{A7}$$

where $\mathbf{I}$ is the identity matrix of $\mathbf{M}$ (thus, M-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:

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

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:

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

where $\varepsilon_s$ 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 $\varepsilon_s$ 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]:

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

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):

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

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.

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