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Original research article

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

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Abstract

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

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

1 Introduction

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

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

How and whether we should evaluate non-marketable ecosystem services is no small debate. Alternative currencies have been put forward, such as energy (McKibben 2007) or happiness (MacKerron 2012), but the decisional power remains in the favour of interests operating in monetary markets (Adamowicz 2004). Non-monetary measures are nevertheless imperative to the sustainable use of ecosystem services as the limits ultimately is biophysical, not economic (Fischer et al. 2007). Advancement of ways to calculate and combine decision metrics in trade-off protocols is therefore gaining research focus (Diaz-Balteiro & Romero 2008; Ostrom 2007; Schlüter et al. 2014). Poff et al. (2010), for example, illustrate a most comprehensive use of compromise programming to aid multi-criteria decision planning by simultaneously optimizing multiple objectives (e.g., plant productivity, biodiversity, streamflow rates, habitat suitability and willingness-to-pay for recreation opportunities). This much-aspired inclusiveness comes with a cost of immense trade-off complexity, which forces that we measure service performances by some kind of normalized indices. Planning participants typically find it difficult to interpret such relative indices (Kangas et al. 2001), and prefer to base their decisions on hands-on measures like biomass or money (but see Adamowicz 2004, p. 439). Along with the ongoing and promising development of multi-criteria analysis (collectively labelled MCA), we advocate to simultaneously explore other ways of implementing trade-off assessment without direct pricing, yet within the ruling scheme of monetary exchange protocols (for a recent review of established and suggested such approaches, see Schuhmann & Mahon 2015).

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

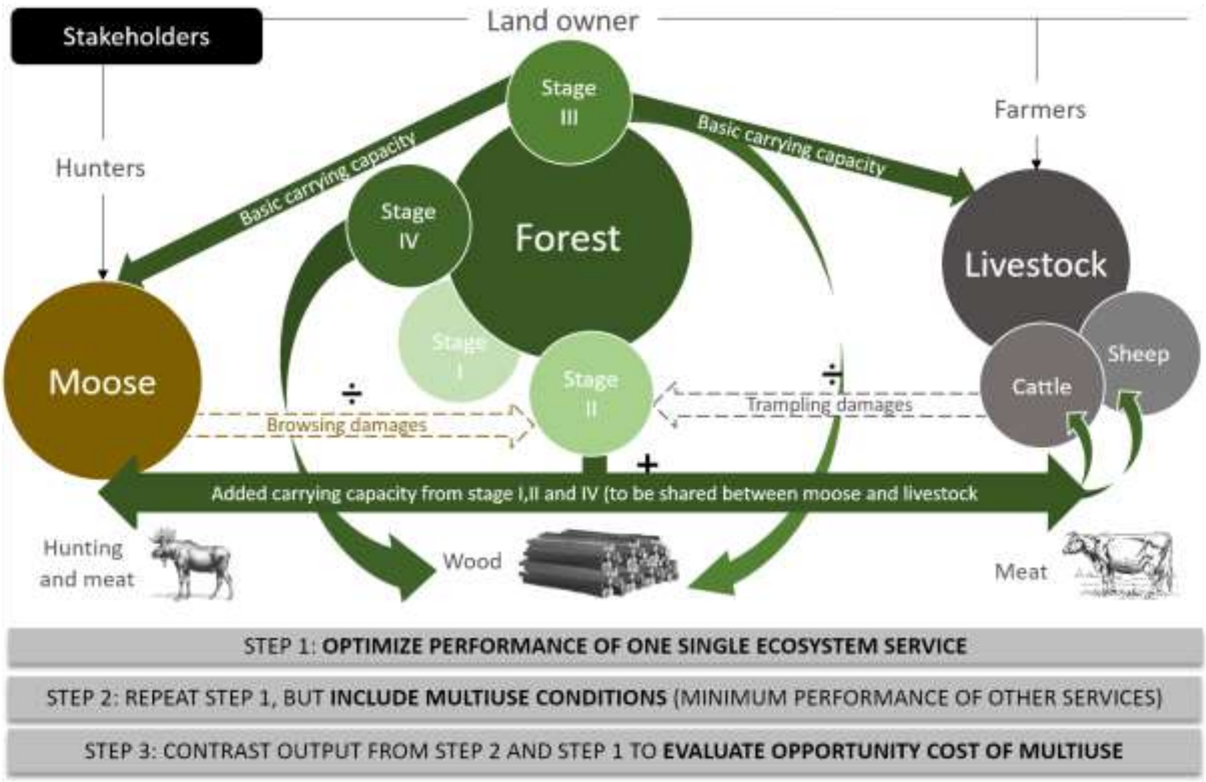

Figure 1. The use of one ecosystem service may both impede and facilitate other services, as partly illustrated above using forest as a case: wood logging in older forest (stage III-IV) substantially contributes to food carrying capacity for moose and livestock, but livestock cause trampling damages 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.

2 Model framework

2.1 Model objectives

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

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

2.2 Model structure

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

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

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

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

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

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

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

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

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

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

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

where pf_s is the net revenue (harvesting costs deducted) (€) per m^3 of wood cut in stratum s, $u_{t,s}$ is the amount of wood (m^3) cut at time t (volumes of trees are stage-specific for a given stratum), cf_s is the fixed cost of conducting one cutting session (e.g., costs of moving equipment between sites, or pre-cutting surveys). Because our model is not spatially explicit, we have to assume that all cutting within a stratum-specific stage represents one cutting session (thus if a stratum is cut in a given year, one unit of cf_s will be deducted). af is the fixed administrative cost of managing the forest. The latter is deducted from the wood income (rather than game or livestock) as forestry normally is the focal interest of landowners in Nordic boreal forests. Forest recruitment after cutting is associated with a cost in spruce forest cr_s (i.e. planting of nursery grown saplings, eq. A11), but not in pine or birch forest (which are recruited by natural seeding). *FEV* is the forest expectation value (see eq. A10):

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

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

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

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

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

In eq. 6, the constant θ is the proportion of new spruce saplings that is trampled each year per cattle-day in the forest. All cattle (cows, heifers and sucklings) are considered to make similar levels of trampling damage. Because even minor trampling damage incurs a severe reduction in future timber quality of spruce, all damaged saplings lose all their monetary value. The proportion of trampled saplings increases both with more cattle-days or with lower proportions of the forest being in stages I and II. The latter occurs because more cattle will then aggregate in these areas, as clearcuts are highly selected habitat for cattle. As for browsed pine, the cost of damaged spruce is corrected with a thinning factor ψ_s (tree density at midpoint stages III and IV / tree density at stage I).

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

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

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

2.3 Model constraints set by non-commodity concerns

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

(i) In line with the ethical notion of sustainability (Leopold 1949), all animal populations must remain below their specific carrying capacity at all times.

(ii) Moose fecundity (as influenced by animal density) must stay ≥ 0.5 calves produced per cow 2+ years. Lower values indicate severe deterioration of health (Solberg et al. 2006). No constraint is set for livestock as their fecundity is determined *ex-situ* by the farming regime, and treated as a constant in the model (Table A.1).

(iii) In line with perceived hunter ethics, moose calves cannot be orphaned by hunters, i.e. the number of hunted cows must not exceed the number of hunted calves divided by the live calf: cow ratio.

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

2.4 Model parameterization and parameter sensitivity

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

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

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

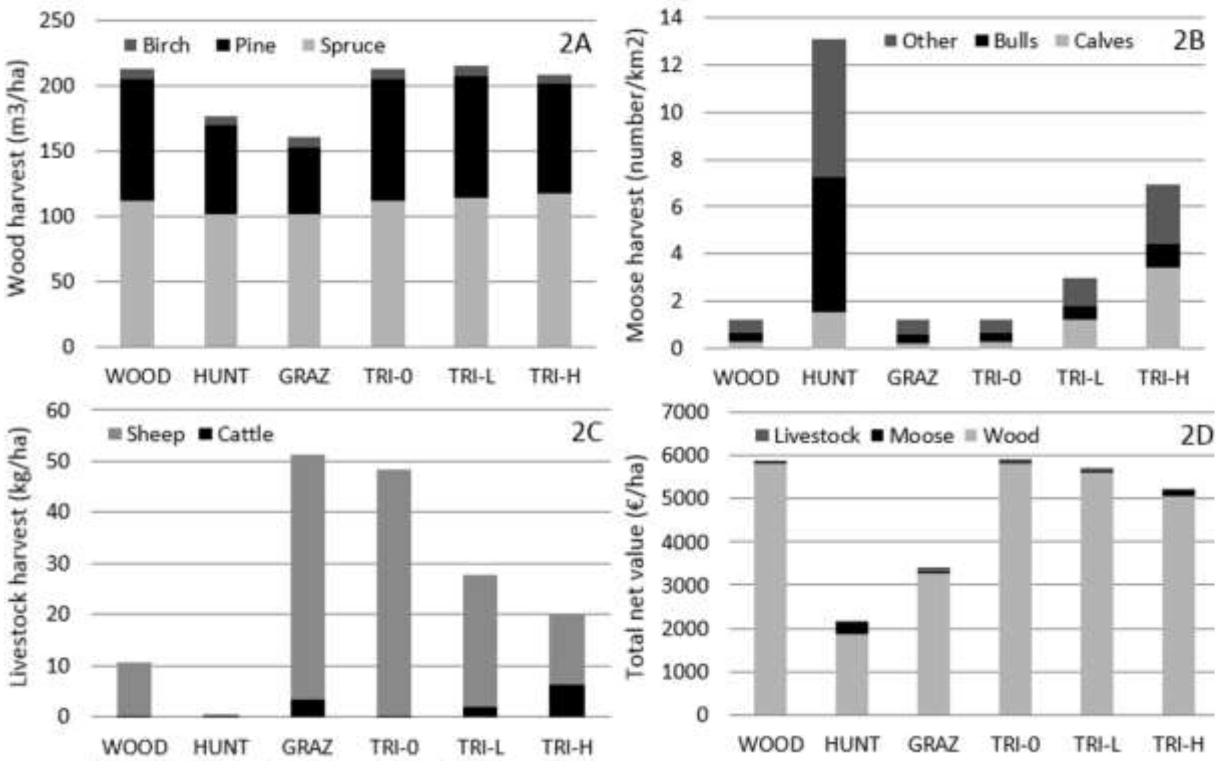

Fig. 2. Potential performance (A-C) and total net present value (D) of forest ecosystem services over 30 years according to a socio-ecologically integrated trade-off model for partly conflicting services, with the objective to maximize net present value from wood production (WOOD), game hunting (HUNT), 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).

3.2 Prioritizing game hunting (HUNT)

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

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

3.3 Prioritizing of livestock grazing (GRAZ)

The optimal strategy when prioritizing livestock grazing (GRAZ) was to eliminate moose (Fig. 2B), maintain the spruce harvest and reduce the pine harvest (Fig. 2A). Livestock had a generally low share of the total net present value potential (Table A.2). Factors facilitating the relative contribution of livestock (G%) to the total net present value were: a higher meat revenue, a higher carrying capacity, a lower Site Index, and higher trampling intensity. Recall that spruce clearcuts were both the main contributor to livestock carrying capacity and subject to livestock trampling damage. Consequently, there were points of inflection in the influence of spruce proportion on livestock relative contribution to net present value (being lower at intermediate spruce dominance). Sheep had a higher income (and meat yield, Fig. 2C) potential than cattle. With all facilitating factors combined, the GRAZ scenario could generate a mean annual net value from sheep of 40 €/ha and 8 €/ha for cattle, compared to 4 €/ha and 3 €/ha with parameters at baseline (sheep and cattle prioritized in separate model runs).

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

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

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

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

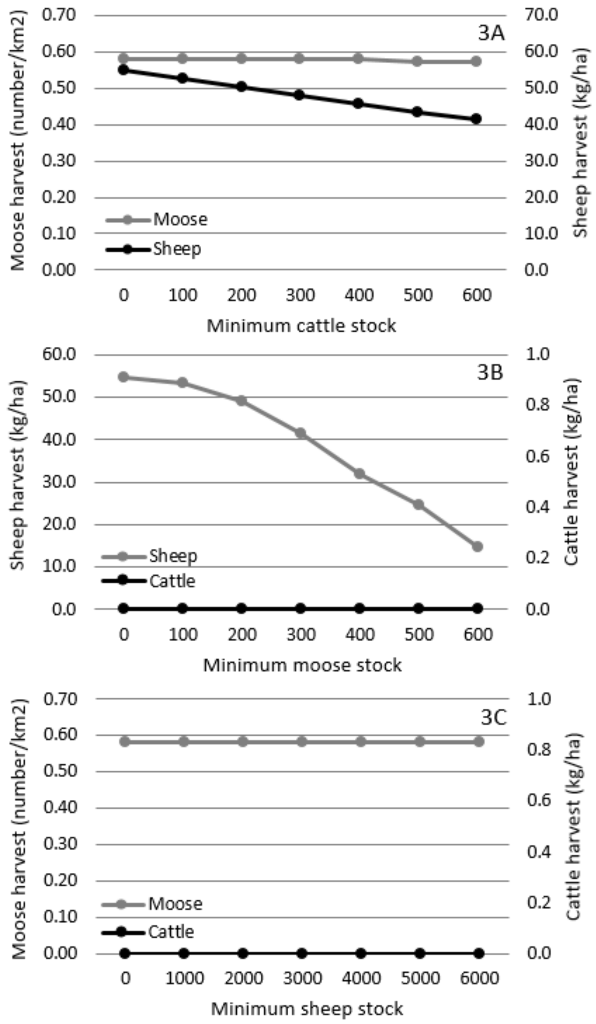

Fig. 3. Potential performance of forest ecosystem services over 30 years according to a socio-ecologically integrated trade-off model for partly conflicting services (wood production, moose hunting and livestock grazing), with the objective to maximize total net present value given various levels of 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).

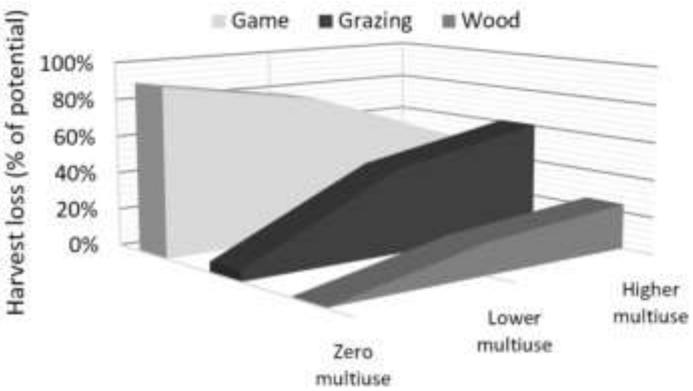

Fig. 4. Loss of potential performance from forest ecosystem services according to a socio-ecologically integrated trade-off model for partly conflicting services (wood production, moose hunting and livestock grazing), with the objective to maximize total net present value given three levels of multiuse 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 m^3 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).

Table 1. Varying parameter values in an optimization model for management of forests with three partly conflicting ecosystem services (wood production, moose 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 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 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 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)					
Parameters	*Baseline*	*Lower*	€/ha (W, H, G %)	*Upper*	€/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)

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

Objective \ Performance	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.

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

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

4 Discussion

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

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

As expected, when we used the compromise programming technique to optimise multi-criteria management of our case system, the unequal weighting of services strongly affected the performance (see also Zekri & Romero 1993). Our case shows that the outcome of a given weighting is not straightforward to predict when density dependent interactions are involved. For example, sheep prioritizing ($w_s = 2$) also gave higher realization of wood potential, because more sheep meant less moose and cattle and therefore reduced damage costs. Likewise, low-level moose prioritizing ($w_m = 2$, but not $w_m = 4$) benefitted cattle, most likely because it facilitated a higher increase in the carrying capacity than the moose could fully consume given the set of other constraints. In a practical application of this sort of resource management, decision-makers must therefore engage in detailed discussions about which weights to be used. In the case of a large forest property, the owner may make the final decision unilaterally according to law. If too little weight is given to less superior stakeholders, the owner may, however, end up in conflict with the local community. To maintain their social capital in the local community owners could probably benefit from compromising somewhat on the net present value (Bowles & Gintis 2002).

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

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

Although we in this study advocate using a quantitative model to aid ecosystem service assessment, we do not argue for the exclusive use of such models. Decision-making regarding the sustainable use of ecosystem services must always be founded in a set of adaptive processes complementing each other (Argent 2009), as there are shortcomings associated with any single model. The scientific and social processes vital to adaptive management can be broadly summarized as: a) Identifying the appropriate spatiotemporal scales of each management option, b) retaining a focus on statistical power and controlled experiments when selecting input data, c) scenario modelling to outline potential outcome of the various management options, d) using model output to synthesize socioecological consensus on the most relevant options, e) evaluating strategic alternatives for achieving these management options, and f) communicating alternatives to the political arena for negotiation and ultimate selection. The link between stages c) and d) is particularly critical (Mapstone et al. 2008), and largely denotes where science ends and politics begin. Without a certain level of stakeholder consensus, the political decisions will be hampered, and if a decision is reached nevertheless, it is bound to exacerbate rather than mitigate conflict (Redpath et al. 2015).

Conclusions

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

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