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Wild Life Conservation in Amboseli, Kenya: Paying for Nonuse Values

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Paying for nonuse values[ψ]

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Abstract: Traditional grazing grounds near Amboseli National Park (Kenya) are rapidly converted to cropland – a process that closes important wildlife corridors. We explore the scope for introducing a "payments for ecosystem services" scheme to compensate pastoralists for spillover benefits associated with forms of land use that are compatible with wildlife conservation. Our results indicate that such a scheme likely enhances global welfare, but that (i) 'leakage' through excessive stocking rates warrant close scrutiny and (ii) that payments increase the risk of overstocking during droughts.

Keywords: paying for ecosystem services, PES, Maasai, elephant conservation, habitat conversion.

ψ An earlier and shorter version of this paper appeared as "Elephants or onions: Paying for nature in Amboseli, Kenya."

1. Introduction

In recent years, a variety of compensation and market-related policies have gained prominence to encourage ecosystem and land managers to change behaviour. While direct financial and market incentive schemes, commonly referred to as direct payments for ecosystem services (PES), now exist in many developed countries experiences in developing countries are limited, with the majority of existing studies focussing on forested watershed and carbon sequestration issues (Landell-Mills and Porras 2002, Wunder 2005).

To date, the PES activities in developing countries addressing watershed issues include "tight" feedback loops where suppliers and demanders are easy to identify (e.g., Pagiola et al. 2003). Less work has been done on payments for global values, especially those *other* than carbon. This is unfortunate because such values may be large, and tapping into them could have far-reaching implications for conservation and development agendas alike.

In this paper, we explore the opportunity to establish an international payment system for non-use values – or cultural values, in the parlance of the Millennium Ecosystem Assessment – associated with wildlife (elephant) conservation near Kenya's Amboseli National Park (NP). Under current trends the long-term future of the Amboseli ecosystem (and its icon – the elephant) looks rather bleak.[1] The objective of this paper is

[1] In addition to the issue of land conversion discussed in this paper, the future of the Amboseli ecosystem may be compromised by the following three factors: (i) the upper forest line on Mount Kilimanjaro shifts down due to an increased frequency and intensity of forest fires, which may have severe repercussions for local climatic and hydrological conditions (for a discussion of the importance of this issue vis-à-vis the more conspicuous issue of the disappearing glacier, see Hemp 2005); (ii) there is a push in the group ranches to sub-divide the communally owned grazing grounds into private plots, which would adversely affect the capacity of the system to support grazing (Boone et al. 2005); and (iii) recently the government of Kenya announced that it would degazette the area from National Park to Game Reserve, and that management would be relegated from the Kenya Wildlife Service (KWS) to the district Maasai Council.

twofold. First, we explore whether efforts to promote elephant conservation near Amboseli NP through a PES scheme represent a viable economic proposition, or not. The outcome of such a comparison may be used to decide whether strategies should be implemented to provide incentives for local households to sustainably manage their rangelands and share this habitat with wildlife. A second, and closely related objective, is to predict how a PES scheme affects conservation (the so-called *additionality* issue) and welfare of the Maasai. To address the second question one would ideally use a household model, but as a fully calibrated Maasai model is not available, we resort to an approximation instead.

The study results are being used to develop a PES project, coordinated by the United Nations Food and Agricultural Organization. The outcomes of the proposed project are threefold: (i) ecosystem-wide management and the development of organizational structures for effective participation and coordination in natural resource management decision-making; (ii) significant increases in wildlife corridors, dispersal areas and habitats through established biodiversity services payments at appropriate sites throughout the ecosystem; and (iii) improved poverty alleviation and household food security outcomes.

The paper is organized as follows. Section 2 provides a brief profile of the Amboseli ecosystem. In section 3, we sketch the bare bones of the SAVANNA and PHEWS models that are used to simulate the impacts, in terms of changes in land use, income and elephant abundance, from a PES system. Section 4 presents the simulation results as well as a rudimentary social cost-benefit analysis. Section 5 concludes.

An integrated, international conservation effort would presumably need to tackle these challenges in tandem. The current study provides a first step.

2. The Amboseli Ecosystem

The Amboseli ecosystem, an area of some 8,000 km^2, comprises part of the Ilkisongo region of southeastern Kajiado District in Kenya and the Longido region of northern Tanzania. Amboseli is typical of African arid rangelands, rainfall is low and unpredictable in time and space. At the heart of the ecosystem is Amboseli NP, the core of a UNESCO Man and the Biosphere Reserve protecting 392 km^2 (about 5%) of the wildlife dispersal area. Amboseli's swamps are fed by subsurface water that percolates though volcanic rock from the forested catchment of Kilimanjaro.

Amboseli NP is fundamental to Kenya's tourist industry, typically ranking second among parks in annual park gate fees – around USD 3.5 million in 2004. In the past, the absence of wide-scale intensive agriculture and the relatively low population density encouraged and provided refuge to a magnificent array of biodiversity, including large and small mammals, birds, reptiles, insects and plants, some of which are rare or threatened. Birdlife International has named Amboseli one of the world's Important Bird Areas.

The Amboseli ecosystem is home to Maasai pastoralists, whose long-practiced livestock activities are well adapted to the variable habitat, and whose land use decisions are a key driver of wildlife abundance in and around the park. However, the majority of Maasai households receive virtually no direct benefits from the wildlife tourism industry. The cash benefits are not distributed fairly nor equally to the landowners (Kellert 2000, Mburu 2003). And the indirect benefits, in the form of reduced school fees, irrigation infrastructure maintenance, livestock sales yards, and other related community goods, often fail to benefit those in most need.[2] The Maasai do bear the costs of managing wildlife habitats, including personal safety, grazing competition, investments to minimize risks, management costs, damage to crops (from eating and trampling), and damage to livestock through the spread of diseases and killing (Norton-Griffiths and Southey 1995, Campbell et al. 2002).

In contrast, the Maasai do receive direct benefits from renting out their land. It is no surprise, therefore, that they have increasingly rented out large areas for irrigated or rain-fed agriculture during the past decade. During the past twenty years, the adjacent areas to the south and east of Amboseli NP (Loitokitok Division) human populations have more than tripled, rain-fed agricultural areas expanded by 3.5 times, and irrigated area increased by 18 times, from around 250 ha to 4800 ha. (Campbell et al. 2003).

Some of the irrigated land was fenced during the late 1990s to protect onions and tomatoes from wildlife, and increasingly those protected croplands impede access to water, food, breeding grounds and to the seasonal migration of wildlife up and down the slopes of Mount Kilimanjaro, and between Amboseli and other protected areas like Tsavo NP.[3] Wildlife populations that had access to all of Amboseli's swamps until the 1970s, now have no access to one swamp and only partial access to three others (Reid et al. 2004). While cropping may be *privately* rational (the returns of cultivation dominate the private returns of wildlife management), it is an open question whether it is also socially

[2] The Maasai communities surrounding the Park are themselves divided about the benefits they obtain from the park (in the form of revenue sharing and job opportunities), and are frustrated that certain beneficial policies that were promised have never been implemented, such as water boreholes outside Amboseli NP. Factions within these communities are dissatisfied with the benefits they obtain, and threaten to intensify pressure on key natural resources in the Park (mainly forage and water) unless they will receive a larger share of the Park's proceeds.

[3] In addition, albeit somewhat beside the main point of this paper, there is evidence that agricultural use of the former grazing grounds is not sustainable because of water pollution, agrochemical use and soil runoff.

beneficial – i.e. what happens when we include eco-services benefiting people outside the Amboseli system in the picture?

3. The Model

The majority of elephants and the other migrating species cannot survive without Amboseli's larger ecosystem, migrating seasonally between the Park and its surroundings. The future of much of Amboseli's wildlife lies in the hands of the people surrounding the Park. Six communally owned group ranches surround the Park. The predominant form of land use has been livestock raising, an activity that is compatible with wildlife conservation. While livestock and wild herbivores may compete for forage, and predators may occasionally kill livestock, historical grazing systems and population pressures seemed to be sustainable and allowed for the co-existence of domesticated and wild animals.

3.1 The Savanna Model

We use a comprehensive and integrative model to represent important processes within semi-arid and arid ecosystems called Savanna, building upon an integrated assessment (Coughenour et al. 2002) of southern Kajiado District (Boone et al. 2005; Thornton et al. 2005). In a drought simulated in such a process-based model, for example, less precipitation falls, evapotranspiration is high, and available soil moisture declines. Plants compete for moisture based on their usage efficiencies, some plants do not receive adequate moisture, and some number of plants die. In a model based on rules, a direct linkage may be made between the severity of a drought and the percentage of plants that will die. It may be that the two models simulate similar mortality in plants, but a process-based model allows for more dynamic and emergent behavior – unanticipated and unique insights are more likely to emerge.

Michael Coughenour started developing SAVANNA in the Turkana region of Kenya more than 20 years ago (Coughenour 1985). Many subsequent improvements and applications around the world have been described (e.g., Coughenour 1992; Buckley et al. 1993; Ludwig et al. 2001; Boone et al. 2002; Christensen et al. 2004; Boone et al. 2004, Thornton et al. 2004; Boone 2005; Boone et al. 2005; Thornton et al. 2005; reviewed in Ellis and Coughenour 1998). A schematic outline of the model is provided in Figure 1.

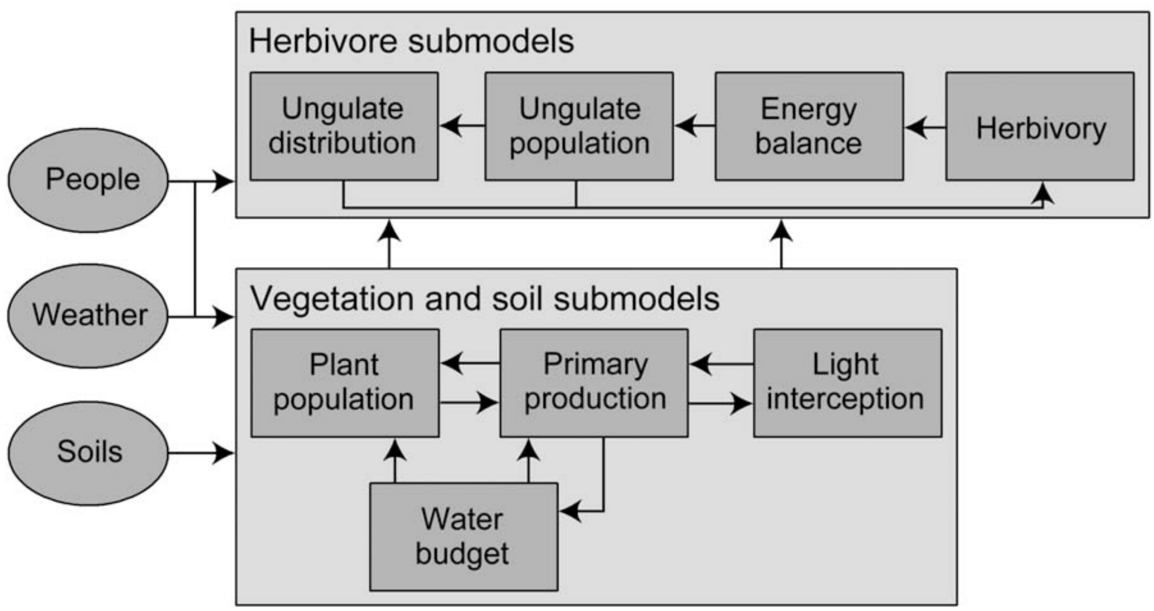

Figure 1: Schematic outline of the SAVANNA model used to simulate the impact of landuse change on wildlife (elephant) abundance.

SAVANNA is a series of FORTRAN computer programs that join to form a spatially explicit ecosystem model that divides landscapes into a grid of square cells. Spatial data layers are used by SAVANNA to characterize the cells as to elevation, slope, aspect, and soil and land cover type. Precipitation data from a series of stations are used by the model to create estimates of rainfall for each cell in each weekly time step. More detailed weather data, include maximum and minimum temperature, plus wind speed, humidity, solar radiation, and CO_2 concentration from a focal weather station is used. Plants are represented by functional groups, such as palatable grass, annual grasses, unpalatable shrubs, and acacias, and distributed on the landscape based on mapped land cover. During simulations, plants compete for water, nutrients, light, and space. Photosynthate is produced and then distributed to plant parts based on allometric equations, such as leaf to shoot ratios. Some portion of photosynthate is allocated to reproductive parts, allowing plant populations to expand in favorable conditions. In any weekly time step, plants may grow, reproduce, be browsed, out-compete other functional groups and expand their population, or die.

Herbivores are represented as functional groups as well, but are often species, such as wildebeest, buffalo, elephants, cattle, and sheep. Animals are distributed on the landscape based on the forage quality and quantity, distance to water, elevation, slope, and woody cover, and temperature. Animals are also distributed using force maps, which capture non-ecological relationships such as if areas area in cultivation and unavailable to ungulates, and are important in scenario analyses. Animals feed on specific plant functional groups and plant parts as reported in the literature (e.g., an elephant will eat more twigs than will a giraffe). SAVANNA is aware of plant and animal heights, for example, so that only plants within reach are eaten. Animals gain energy from the forage they eat, and use energy for basal metabolism, gestation, lactation, and travel. Surplus energy goes to weight gain, reflected in reported condition indices. Mortality occurs in each time step, and reproduction may occur in appropriate months, with animal cohorts tracked in a Leslie matrix (Leslie 1945). Birth and mortality rates are tied to animal condition indices, so that birth rates decrease and mortality increases as condition indices decline, one of many potential feedbacks within the model. SAVANNA is generally used on landscapes from 500 to 20,000 km^2, and in simulations that span from 10 to 100 or more years (Ellis and Coughenour 1998), and produces charts and maps at monthly intervals (e.g., Boone et al. 2002).

The SAVANNA ecosystem model has recently been used in a series of analyses in East African conservation areas exploring the balances between pastoralists, their livestock, and wildlife in management decisions. We conducted an integrated assessment in Ngorongoro Conservation Area, Tanzania, that explored ecosystem effects from policy changes affecting access, water availability, veterinary care, cultivation, and human population growth (Boone et al. 2002). In a follow-up project, the Conservator of the area asked us for more detailed analyses relevant to tradeoffs between livestock and wildlife, cultivation, and other issues. Fifteen methods were used to judge the capacity of the area to support herbivores, and ecosystem modeling suggested the area was ca. 15% below its capacity to support herbivores, limited by disease. Humans had 6 TLUs/AE (tropical livestock units per adult equivalent) in 1991, and by 2000 that number had dropped to 3 TLUs/AE – livestock numbers remain similar, but human population is increasing. Our modeling confirmed that it is not possible for the residents of the area to return to 6 TLUs/AE; that number of livestock cannot be supported. We created the first high-resolution map of cultivation in Ngorongoro, yielding 9803 ac (3967 ha) of cultivated lands, then modeled cultivation from 0 ac to 50,000 ac, exploring effects on resident wildlife, livestock, and human wellbeing. We did not find significant changes in wild or domestic herbivore populations in response to increasing cultivation, but the benefits to Maasai were profound (Boone et al., in press). Our research shows that small-scale cultivation in Ngorongoro, in its current configuration, is not a significant threat to wild ungulates.

Another study focused on the effects of group ranch subdivisions. In southern Kajiado District, Kenya, lands were subdivided from intact sections into group ranches, and now are being divided into 60-100 acres (24-40 ha) parcels used by individuals. Animals confined to parcels have few options to reach ephemeral forage patches. Integrated assessments suggested that, even with access to water retained, subdivision to individual parcels can lead to large declines in livestock. In Eselenkei Group Ranch within Amboseli Basin, subdivision to 1 km^2 parcels led to a 25% decline in livestock that could be supported, relative to the intact group ranch (Boone et al. 2005). In more productive Osilalei Group Ranch, livestock populations did not decline under subdivision. We hypothesize a uni-modal relationship, where areas of very low or very high productivity and landscape heterogeneity are not strongly affected by fragmentation, but areas of intermediate productivity are sensitive to heterogeneity. Results from PHEWS (see below) confirmed that declines in livestock populations have profound negative effects on the wellbeing of Maasai.

The Savanna model applied to southern Kajiado District was used in modeling here, to analyze the consequences of introducing a PES scheme. In the application, seven plant functional groups are captured in the case study's SAVANNA model: palatable grasses, palatable forbs, unpalatable grasses and forbs, papyrus (*Cyperus papyrus*) swamps, palatable shrubs, unpalatable shrubs, and deciduous woodlands. Nine animal groups were modeled: three livestock species (cattle, goats and sheep), and six wildlife groups (wildebeest, *Connochaetes taurinus*, zebra, *Equus burchellii*, African buffalo, *Syncerus caffer*, grazing antelope, browsing antelope, and elephants). See Boone et al. (2005) for species comprising grazing and browsing antelope groups. A variety of data sources were used to parameterize the application for southern Kajiado District, described in Boone et al. (2005), including examples of literature used. The ecosystem model was calibrated using sources such as an net primary production database (Kinyamario 1996), satellite imagery, which relates well to regional stocking levels (Oesterheld et al. 1998), and information from important literature sources (e.g., Bekure et al. 1990; De Leeuw et al. 1998).

3.2 The PHEWS Model

In an ideal world the SAVANNA model would be linked to a process-based, detailed and fully calibrated household model that captured the myriad of response Maasai may have to changing circumstances. Such a model would combine the preferences of the Maasai with respect to goods and services they consume (including their utility from leisure) with a set of constraints – a budget constraint, time constraint, production possibilities, etc. However, such a model is not available for the study area. Instead, we use an approximation of such a model, calibrated for pastoral households in East Africa, called PHEWS (Pastoral Household Economic Welfare Simulator – see Thornton et al. 2003 for details).

PHEWS is not a conventional utility maximizing model, but instead is based on a set of rules that households follow when trying to secure caloric intake. It is well known that rainfall and income from herding are highly volatile in this part of the world. PHEWS keeps track of dietary energy flows and prescribes a certain series of actions when intake falls short of a desired level. In addition to a target caloric intake, the pastoral households also have specific target levels for their livestock – preferred numbers of heads for goat and cattle stocks. Pastoralists are assumed to use livestock as a buffer in periods when household income and consumption are low, and invest in livestock (if the stock is below the target level) when income is high and caloric requirements are easily satisfied. Remaining funds are placed in a so-called "cash box" where it is stored for future use when income is low (Thornton et al. 2006).

When caloric intake from consumption of animal products, maize and sugar is insufficient to meet the threshold, the household tries to use its "cash box" (if available) to make up for the deficit. If this fails it sells a goat or cow. If all fails, the model assumes that there will be outside relief from some exogenous source. For this reason the model is not particularly useful for capturing Malthusian population dynamics, say, and we simply assume that the human population is constant.[4] The model is designed to be tightly linked with SAVANNA, so that the two components exchange information at each time step. SAVANNA passes livestock numbers and climatic information to PHEWS, which is used in decision-making about livestock and crop sales. In turn, PHEWS passes changes in livestock numbers back to SAVANNA, to keep accurate accounting of herds. TLU/AE, the proportion of needs met with their own food, livestock purchases, and supplemental needs are important outputs in analyses.

[4] In reality of course the population is not constant. It may change because of natural population growth and mortality, but also because of migration patterns. It is possible that both replenishment and migration react endogenously to implementation of a PES scheme. While we ignore this in the analysis, this is somewhat that should be considered when actually transferring money.

3.3 Three scenarios

We distinguish between three different scenarios, which are compared to highlight effects of different management practices. In control scenario A we consider the base case where parts of the group ranches that surround the park are converted to fenced in cropland (but note that we assume that the fenced in areas were used for cropping throughout the entire study period – from 1977 to 2000 – and that in reality fencing only started in the 1990s). We use historical rainfall patterns to simulate livestock and wildlife abundance over time and space. In the pastoral scenario B we explore the case where the fenced in area is returned to grazing ground and accessible for wildlife and livestock alike. One may think of this as a command-and-control approach to conservation, simply banning the use of fences. We simulate the impact on wildlife and livestock, but also on Maasai income. Finally, in PES scenario C we consider what happens if we compensate the Maasai for restoring the grazing grounds. That is, in return for giving up the privately profitable option to rent out land to onion growers, the Maasai are assumed to engage in an easement deal with a funding agency that offers a competitive rate of return on the land. Compared to scenario B the Maasai budget constraint is therefore relaxed, which means that households are better able to meet their target consumption and livestock levels.[5]

This approach involves comparisons between simulations where the only attributes changed were areas available for grazing and payments to Maasai. The model was parameterized to agree with current conditions to the degree possible, but the approach is not predicated on responses being absolutely correct, but rather on

[5] In reality a fourth scenario is being discussed: the case where fences are not removed but where agriculture *outside* the fences is controlled to enable a free flow of animals between areas. In theory we could readily analyse this case, but the resolution of the current model is too coarse to yield reliable results. The scenarios considered in this paper are more dramatic cases, sharply illustrating the main tradeoffs.

comparisons between simulations that are otherwise parameterized identically. Our results are not intended to provide precise predictions about how the elephant population may change in the future; too many unforeseen circumstances may affect the trajectory. Rather, we provide examples of tradeoffs associated with PES systems, and identify the direction and magnitudes of change in wild and domestic ungulates, and in Maasai well-being.

4. Simulation results and CBA

In this section we present the simulation results of the three scenarios, and we use these results as input in a cost benefit analysis. We try to address the question whether a PES scheme for elephant conservation is welfare enhancing at the global scale, or not. We also use the output to discuss the form that transfers from conservationists to pastoral households may take.

4.1 Returning cropland to range land

Figure 2 summarizes the impact of returning the fenced-in cropland to grazing grounds on elephant abundance. The dashed upper line reflects elephant abundance in scenario B (no fences) and the solid lower line reflects the number of elephants in control scenario A. The figure also shows the historical pattern of rainfall in the study area (light dashed line).

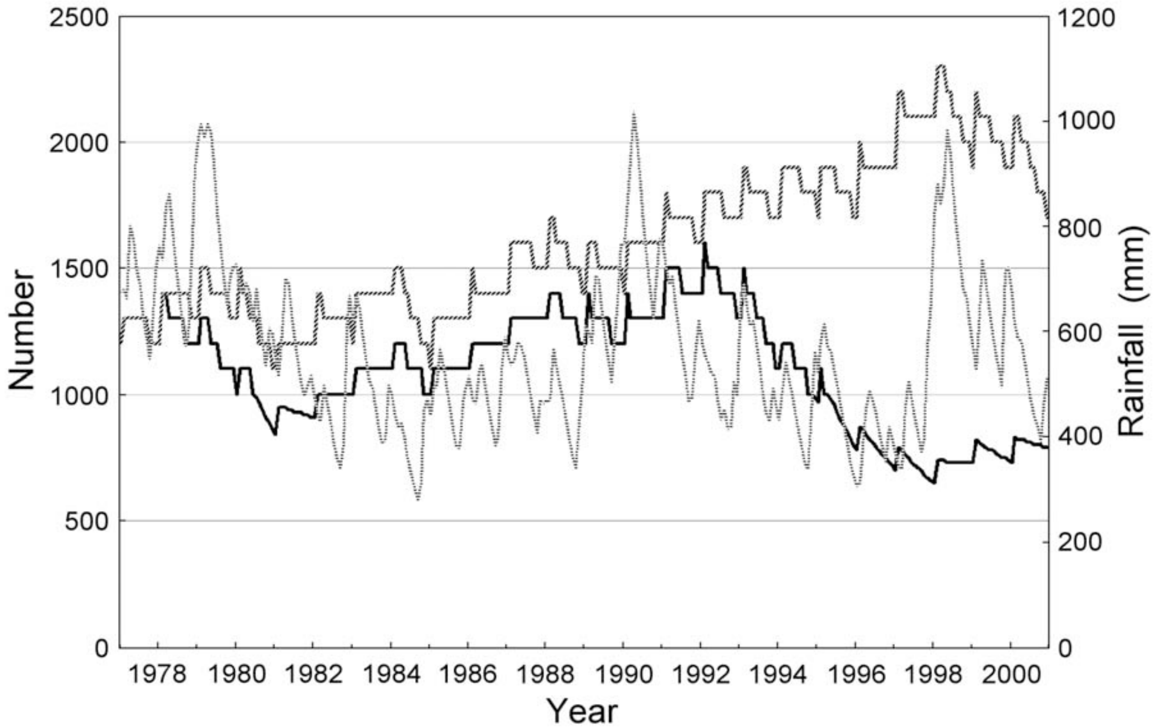

Figure 2: Elephant abundance with (solid) and without (heavy dashed) habitat conversion. Precipitation over 12 months (light dashed) is overlayed for comparison.

Not surprisingly, expanding elephant habitat translates into a larger number of elephants. However, during the first 15 years of the simulation exercise the impact is very modest – typically in the range of only 100 to 300 extra elephants per year, or a modest 15% increase in abundance. It appears as if the PES scheme is hardly worthwhile. But the situation abruptly changes after 1992, when stocking rates are rather high and a serious drought hits the area. The elephant population in the control scenario collapses to about 50% of its pre-drought level of abundance while the elephant population in the pastoral scenario increases. Considering the entire study period from 1977-2000, the *average* number of elephants in the pastoral scenario is about 500 head larger than in the control scenario. But averages tell only part of the story: the main benefit from removing fences is that the elephant population is much more resilient to changes in (environmental) conditions when it has access to a wider set of base resources.

The interpretation of these results is as follows. In times of sufficient rainfall, the swamp areas converted to cropland do not represent a key resource for elephants. Opening up these areas implies they have access to more food, so we observe a modest increase in the population. However, the picture changes in times of drought, when access to the swamps for food and water becomes necessary to support the elephants. If this access is denied, water and food become critical factors and the population crashes.

The elephant population in pastoral scenario B increases amidst the drought of the mid 1990s because it faces less competition from livestock. Faced with a drought, the Maasai have no option but to sell part of their large stock to support their families, to buy grain and more drought-resistant small stock. The loss of milk from the large stock demands more large animal sales, which in turn means less milk, etc., in the downward spiral seen here and sometimes seen in Maasai communities. In the simulation goats came to dominate herd composition. This represents a fundamental tradeoff of the command-and-control option to conservation: if it is effective at promoting elephant conservation by restricting the Maasai's use rights of the swamps, the costs of this "success" are borne entirely by the Maasai who see their herds shrink and income position deteriorate. Since most of the non-use values associated with conservation are transboundary, this is clearly unfair.

4.2 The effect of paying for ecosystem services

Figure 3 summarizes the consequences of establishing a PES system, where the Maasai lease their cropland to a conservation agency (as opposed to onion farmers), and where the restored grazing grounds and swamps are available for livestock and wildlife. The upper solid line represents the elephant population when a PES system is in place – the scenario C – and the lower dashed solid line, again, depicts pastoral scenario B discussed above, where fences have been removed but where no compensation takes place.

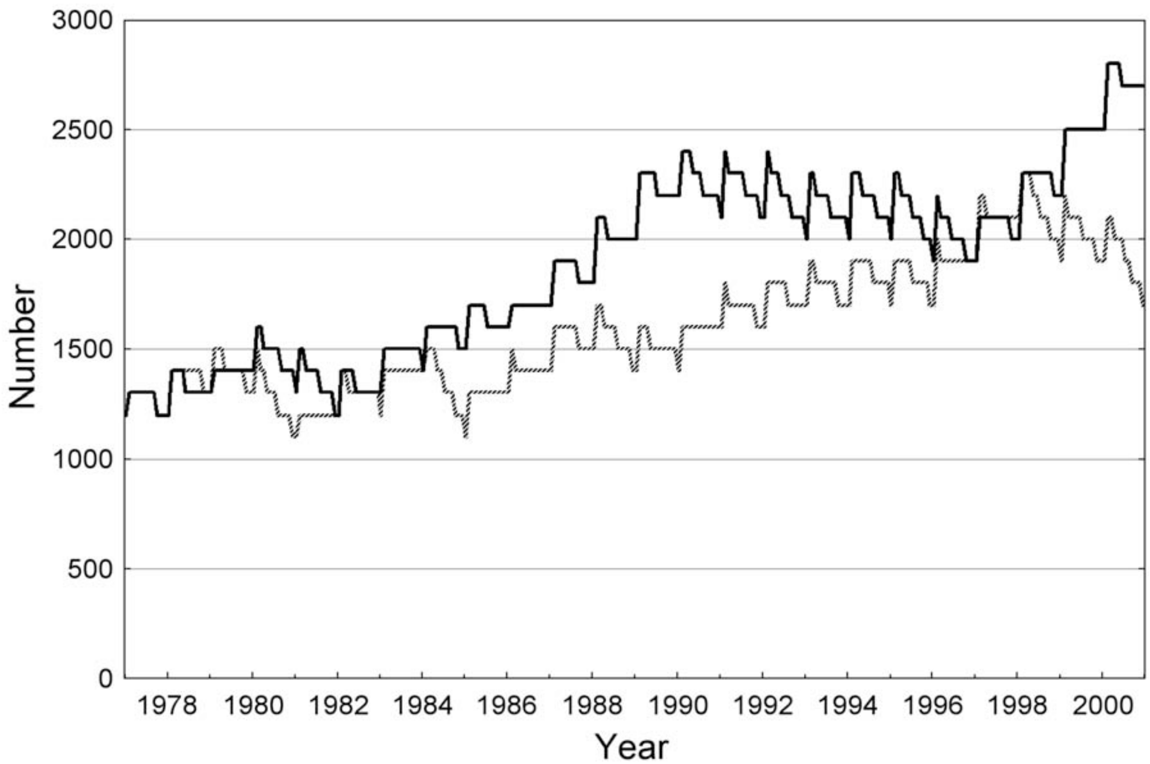

Figure 3: The effect of paying Maasai for not renting out their lands on the elephant population [a comparison of pastoral systems with (solid) and without compensation (hatched)].

The first thing to notice is that a fair transfer to the Maasai did not compromise elephant conservation – the opposite is true. Key resource areas and other rangelands remained available because of the PES agreements limiting cultivation. Elephant numbers exceeded those when the entire area is pastoral because the transfer enabled the Maasai to support a livestock herd that was close to the preferred size and composition.

While increasing livestock herd size is detrimental for conservation – livestock and wildlife compete for base resources – the same is not true for the changes in composition brought about by the PES system. Specifically, cattle diets overlap less with elephants than do goat diets. The PES system enabled the Maasai to gradually expand their cattle stock (towards a herd that exceeds the herd under pure pastoralism by some 4.000 heads, or an increase of some 25% relative to the pastoral scenario B), and move away from goats. In the final periods of the simulation exercise, the goat herd under scenario C is some 10.000 head smaller than in pastoral scenario B (representing a 33% reduction). Because goats and elephants have overlapping diets – they compete to some degree for food – this change in the composition induced by a relaxed budget constraint favored elephants.[6]

The main insight is that poverty alleviation and conservation may go hand-in-hand. Implementation of a PES scheme will both make the Maasai better off (in our specification: they are fully compensated for the foregone returns from leasing out their land, and as a bonus they can use the restored grazing grounds for their own livestock), and will enhance and stabilize elephant populations. The lack of a tradeoff follows from ecological interactions between species – a feature that is perhaps easily overlooked by economists. Capturing such interactions implies developing multi-disciplinary models such as the one advanced here.

4.3 First attempt at a (partial) cost-benefit analysis

The observation that the PES scheme makes elephants and Maasai better off does not necessarily imply that it is welfare enhancing, because there are costs to consider as well.

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[6] By the same token: Note that the change in livestock composition from goats to cattle will adversely impact grazing species of wildlife that compete for food with cows.

How do the costs and benefits compare? A full cost-benefit analysis may account for the distributional consequences (giving extra weight to income of the Maasai) and should account for transaction costs, etc. In this partial cost-benefit analysis we ignore these issues and focus instead on a more narrow question: does the conservation value created by the PES exceed or fall short of the opportunity costs of conservation – the foregone returns to cultivating onions, proxied by the rental payments to Maasai?

Upon comparing control scenario A with PES scenario C, the PES scheme produces benefits of some additional 600-700 elephants per year (average value). How much does the international community value the conservation of some 650 elephants? Answering this question is not easy. First off, we would be interested in marginal values and this information is not available to our knowledge. Second, the appreciation of elephants is income and location specific: elephants are likely a normal good (or perhaps even a luxury good) in the sense that demand for them increases with income. And geography matters because elephants are a real threat to the safety of people who live with them (41% of villagers polled in Cameroon wanted elephants removed or fenced in, and a significant minority wanted them shot – see WWF 2000). When considering the non-use value of charismatic species like elephants it is not obvious which reference population should be included in the aggregation exercise.

Because of the uncertainties that inevitably surround point estimates of the value of elephants we turn the question around: focus on the costs first, and then argue whether it is plausible that aggregate values are sufficiently large to overcome these costs or not. Needless to say we have a fairly decent handle on the (opportunity) costs of the PES scheme. Based on observations in the field we use a payment of KES 10,000 per acre per year (about \$10), so multiplying the fee by the relevant area of cropland yields a total cost of \$112,500 per year. Assuming constant marginal cost, this translates into a cost of some \$175 per elephant per year (divide by 650).

Assuming that the marginal value of elephant conservation is constant (a strong assumption), a prerequisite for the PES scheme to be globally welfare enhancing is that households in Europe and the United States are willing to pay \$0.60 per year for African elephant conservation.[7] Of course it is an open question whether households are indeed willing to pay such amounts, but evidence gleaned from contingent valuation studies into the willingness to pay for other species (for an overview, see Loomis and White 1996) suggests that this number is not excessive. One specific study aimed at valuing Asian elephants (in Sri Lanka) also produced an estimate of WTP amongst the people of Sri Lanka that would have been sufficiently high – some \$12 per household per year (Bandara and Tisdell 2005). We conclude that a PES effort for the Amboseli region is likely to make good economic sense.

4.4 Exploring leakage

In this section we explore how robust these results are with respect to alternative specifications of Maasai behavior. The PHEWS model is based on the assumption that pastoralists use PES funds to re-balance the composition of their livestock herd (purchasing extra cattle at the expense of goats and sheep), and store some of the money in their cash box for future use. What happens if, instead, *all* the new funds are used to

[7] The calculation is as follows. Current estimates of the African elephant population amount to some 500.000 head (Blanc et al. 2003). Assuming a minimum benchmark cost of \$175 per elephant per year, the total benefits of elephant conservation should amount to $\$87.5\times10^6$ per year. Dividing by the number of households (150×10^6) this amounts to \$0.60 per household per year.

purchase additional livestock in the *same* proportion as current livestock holdings? We have used SAVANNA to explore this issue.

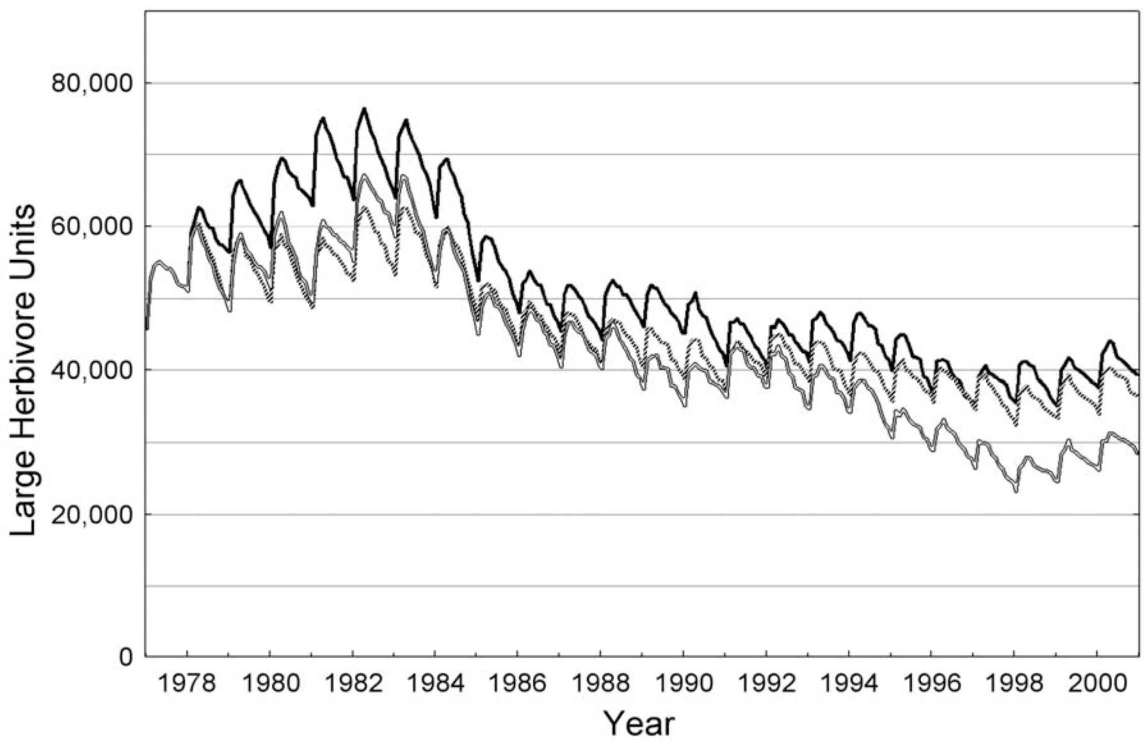

Figure 4: The Leakage effect. Wildlife stocks in the presence (solid) and absence (open) of payments for ecosystem services, as well as if payments are used entirely to purchase livestock (dashed).

Representative results are provided in Figure 4, depicting wildlife populations for three different scenarios: (i) PES payments going to households (solid line), which is just scenario C based on the PHEWS model, (ii) PES based on the assumption that all money is immediately converted into livestock (dashed line), and (iii) the control scenario A above (open line). The curves for scenario's A and C are different than the ones depicted in Figures 2 and 3 because they are based on an aggregate measure of wildlife – they contain, but are not limited to, elephant abundance. Two results follow from Figure 4. First, it is clear (and unsurprising) that the conservation effects of PES are attenuated when the Maasai convert all payments into livestock – the dashed curve is below the solid curve. Livestock demand for forage exceeds the carrying capacity by some 20%, and overgrazing and competition for food forces the wildlife population down. In particular smaller-bodied herbivores showed such compensatory changes in abundance in response to a rapid increase in livestock stocking (elephants are less sensitive).

Second, and more interestingly, upon comparing the new scenario where PES payments are used to buy livestock to the control scenario without PES it is evident that it is difficult to unambiguously rank the scenarios in terms of conservation effects. There are periods where the wildlife populations with PES are smaller than those occurring in the control case with farming and fences. Throughout the 1990s this situation reverses, and the conservation effects of PES are positive. The reason for the ambiguity is that PES pushes both the extensive and intensive margin of herding. The extensive margin is pushed out as more rangeland is made available, but the intensive margin shifts simultaneously as Maasai increase their stocking rates. The net effect on the availability of food for wildlife is ambiguous, but will be determined by the *relative price* of livestock. If this price is high (relative to the PES payment) pastoralists respond by modestly increasing their stocking rates, and the extensive margin effect dominates. However, as the livestock price becomes sufficiently low (or as the payments translates into a sufficiently large number of new livestock heads), the gains from extra rangeland are dissipated through the losses from extra competition for food. In the absence of information on relative prices (context-specific) and a better understanding of the pastoralists' objective function it is hard to predict the outcome of PES systems. This is an area worthy of more research.

4.5 Destitution and droughts: Smoothing incomes

An important advantage of PES, according to several proponents, is its role as a potential income-smoothing device. While incomes of agriculturalists and pastoralists vary with the vagaries of nature – droughts, wildlife damages, etc. – PES promises a (relatively) stable flow of income, enhancing welfare for risk averse households or households close to the poverty margin. We explore this issue in more detail with the SAVANNA/PHEWS modeling system.

We "hardwired" the weather pattern by artificially adding droughts to the system,[8] and explored the impact of implementing PES on (i) the amount of gifts or relief required by households (the last resort to avoid starvation according to the PHEWS model), and (ii) the amount of emergency sales of livestock by the Maasai (arguably a measure of resilience). Annual rainfall in Kajiado is some 550 mm (averaged across the site and over the study period), and we define a drought by the average rainfall minus one standard deviation: 550 – 206 = 344 mm or less. We created several weather patterns, with varying drought intensities, but here we only report the outcomes of the most extreme weather pattern. This is a pattern with two drought periods, of two years each, during 1986-1988 and again during 1992-1994 – details of the other weather patterns are available upon request (Figure 5). We use this weather pattern again in simulations of scenarios A, B and C.

[8] Since the response in any given year is dependent upon the history of the ecosystem to that point, comparisons across years are always a problem. Therefore it is appropriate to alter weather patterns, run analyses with and without PES in place, and then compare the responses during those effects, plus integrated over the entire period modelled.

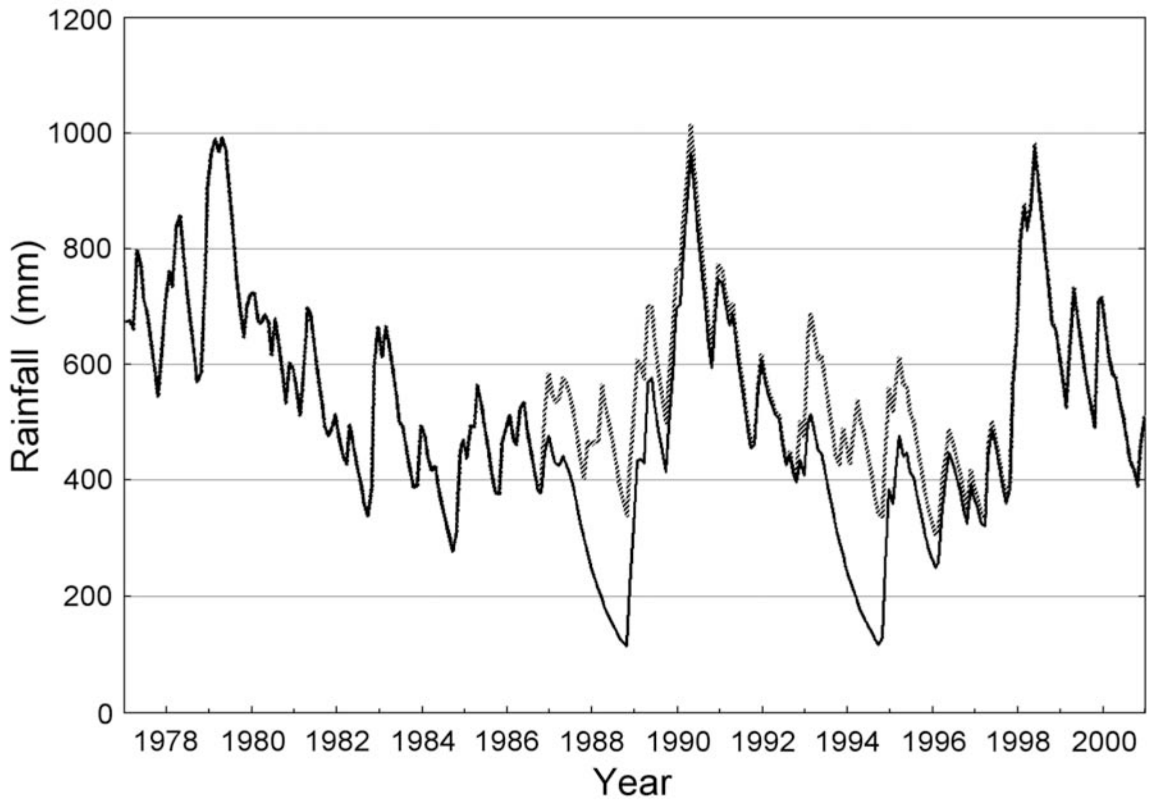

Figure 5. Droughts created in the weather record, 1986-1988 and 1992-1994 are shown (solid), relative to the observed rainfall pattern (dashed). Values shown are 12-month running-means of rainfall.

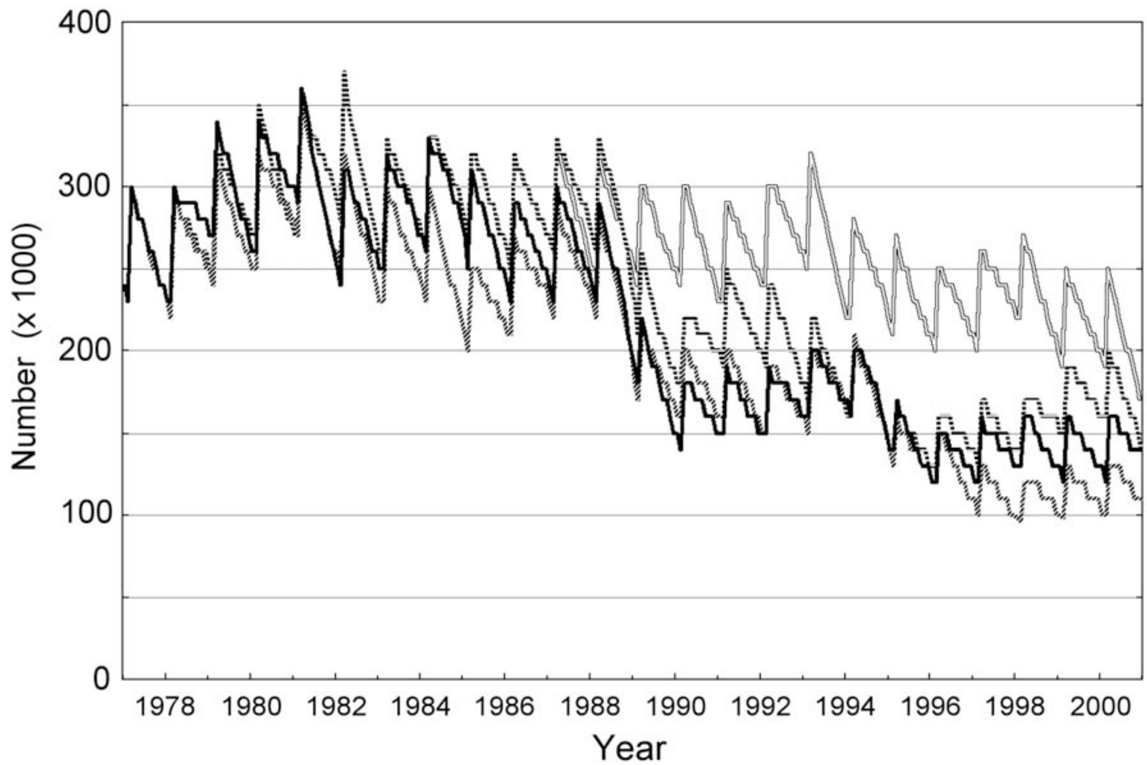

Figure 6. Effects of two-year droughts in 1986-1988 and 1992-1994 on cattle populations in the presence (solid) and absence (hatched) of payments for ecosystem services, as well as when cultivation was in place and payments were not made (horizontal dashed). For comparison, we show the population trend when cultivation is in place and observed precipitation is used in the simulation (open line).

One key result is reported in Figure 6, showing how droughts and PES interact and impact on cattle stocks in the study region. It is obvious that droughts adversely impact livestock numbers. As a point of reference we have added the original scenario A with *actual* rainfall (i.e. not hardwired for droughts) to the Figure, and it is evident that the two droughts depress cattle numbers below the original scenario. Interestingly, the impact of drought is context dependent. While the first drought affects scenarios with and without PES rather similarly, the second drought implies that households in scenario B (no PES) have to sell part of their livestock to sustain themselves. Hence, unsurprisingly perhaps, there are conditions where PES leads to larger cattle populations.

The main results of the drought scenario are threefold. First, it appears that the impact of PES on emergency relief reliance is modest. Most herders in the Amboseli ecosystem have access to a significant capital stock from which they can draw in times of hardship – their livestock herd. So, while a PES system implies and modeling suggests that households have to resort to emergency sales less frequently, it actually has little effect on their ability to sustain themselves. Specifically, we find that PES does not reduce the need for relief relative to scenario A (the case with cultivation), but has a small impact relative to scenario B, the case without cultivation and without PES, where some 8% of the poorest households require relief during extreme droughts.

Second, and closely related to the previous point, because PES affects the magnitude of emergency sales it also affects grazing intensity and, hence, via competition for forage it also impacts on wildlife abundance. PES payments not only allow households to retain more livestock over the long term (e.g., 3.61 TLUs per adult equivalent with PES, 3.32 TLUs/AE without, and 3.73 TLUs/AE with cultivation in place), they also impact on the timing of sales in response to environmental conditions. As explained earlier, sales of cattle are typically accompanied by purchases of goats or sheep, hence according to PHEWS the composition of the livestock herd changes as PES is implemented. Wildlife species that compete for resources with cattle (goats) will therefore lose (gain) following introduction of a PES scheme. Echoing our earlier findings, elephants are better off – access to the swamps through the removal of cultivation and reduced competition through a "more favorable" composition of the livestock herd (i.e. less goats) increases their population by ca. 400 animals.

Third, since the conservation effects of PES depend on the behavioral prescriptions of the Maasai, the actual impact could be more detrimental than suggested by the modeling results if PHEWS underestimates the stocking response of the Maasai. In section 4.4 we showed that PES could result in significant leakage effects if Maasai would use all extra income to purchase additional livestock. These adverse outcomes are amplified by severe droughts. If PES enables the Maasai to maintain livestock populations in the face of extreme stress, the result is more extreme rangeland degradation (with adverse consequences for wildlife and livestock alike). Taken together, these findings cause us to qualify the high expectations of PES for income stabilization somewhat – income smoothing comes at a cost in our pastoral system.

5. Discussion and conclusion

In this paper we explored the opportunities for implementing a payment for ecosystem services (PES) scheme on Maasai group ranches near Amboseli NP. Wildlife migrates seasonally in and out of the park, and conserving wildlife in a sustainable fashion implies securing land use types outside the reserve that are compatible with wildlife. Livestock grazing is an example of such a compatible land use type. Fenced in cropping is not. Due to the many and potentially complex interlinkages between human and natural systems it is imperative to analyze these issues with a model that integrates insights from ecology and economics.

PES is an increasingly popular instrument for promoting conservation, especially in developed countries. In recent years, PES has been introduced in developing countries, in particular in the context of watershed management and carbon storage.

However there is no reason to discount the potential use of PES as a mechanism to align potentially opposing interests in the area of wildlife management or biodiversity conservation (areas where non-tangible non-use values are likely important – spilling over national boundaries). We conclude that PES may be a powerful tool in the Amboseli ecosystem because it promotes conservation and contributes to alleviation of poverty. Moreover, and interestingly, the basic behavioral model that we employ (PHEWS) suggests that these beneficial effects seem to mutually enforce each other: there is no tradeoff between making the Maasai less poor and protecting elephants. Our analysis also indicates that the proposed PES scheme enhances global welfare. An important caveat is the potential issue of 'leakage' or 'slippage'. If we use a simple mechanical rule to describe Maasai behavior (i.e. 'use all extra funds to purchase extra livestock'), then much of the gains from habitat expansion are dissipated through extra competition for food between livestock and wildlife. The risk of rangeland degradation is particularly hazardous in the context of a severe drought – income stabilization might trigger massive range degradation if herders are not forced to sell cattle in times of drought. This is clearly an issue that needs to be explored in more detail.

One final issue remains – how should the PES project be funded? In light of the very significant non-use values associated with elephant conservation it seems appropriate to turn to funding opportunities like the Global Environmental Fund. However, GEF only funds projects for a period of five years, and afterwards projects should be self-sustaining. This appears a shortsighted policy in the case of conserving ecosystem services for non-use values. Such services represent ongoing flows of benefits that accrue to the world population as a public good, and in the absence of sustained compensation such flows will eventually be curtailed. Unlike the case of watershed management it is hard to identify parties with a strong interest in privately 'purchasing' the service – making them available for the whole world at zero cost. The GEF could play an important coordinating role in this respect, and should strive for sustained payments in that case.

Fortunately matters need not be so complex for the case of the Amboseli ecosystem, which is a very popular tourist destination. With 200.000 tourist days a year, the PES program could be easily funded with a relatively minor increase in the Park entrance fee – from US$30 to US31 – or with the introduction of a modest bed tax. Having visitors pay for conservation implies that non-visitors are free riding, and receive their non-use values at zero cost. Clearly such free rides are not always feasible elsewhere, and we therefore recommend the establishment of a new institution – a revamped GEF or otherwise – that will be able to collect payments for nonuse values and channel them to those areas in the world where these values are supplied.

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References

Blanc, J.J., C.R. Thouless, J.A. Hart, H.T. Dublin, I. Douglas-Hamilton, C.G.. Craig and R.F.W. Barnes, 2003. *African Elephant Status Report 2002. An Update from the*

African Elephant Database. Occasional Paper of the IUCN Species Survival Comm. No. 29. Gland, Switzerland and Cambridge, UK: IUCN, UNEP. 304pp.

Bekure, S., P. de Leeuw, B. Grandin, and P. Neate (EDS.). 1991. Maasai herding: analysis of the livestock production system of Maasai Pastoralists in eastern Kajiado District, Kenya. Addis Ababa, Ethiopia: ILCA, ILCA Systems Study 4.

Boone, R.B., M.B. Coughenour, K.A. Galvin, and J.E. Ellis. 2002. Addressing management questions for Ngorongoro Conservation Area using the Savanna Modeling System. *African Journal of Ecology* 40:138-150.

Boone, R.B., K.A. Galvin, M.B. Coughenour, J.W. Hudson, P.J. Weisberg, C.H. Vogel, and J.E. Ellis. 2004. Ecosystem modeling adds value to a South African climate forecast. *Climatic Change* 64:317-340.

Boone, R.B. 2005. Quantifying changes in vegetation in shrinking grazing areas. *Conservation and Society* 3:150-173.

Boone, R.B., S.B. BurnSilver, P.K. Thornton, J.S. Worden, and K.A. Galvin, 2005. Quantifying declines in livestock due to land subdivision. *Rangeland Ecology and Management* 58: 523-532.

Boone, R. B., K. A. Galvin, P. K. Thornton, D. M. Swift, and M. B. Coughenour. In press. Cultivation and conservation in Ngorongoro Conservation Area, Tanzania. Human Ecology

Buckley, D.J., Coughenour, M.B., Blyth, C.B., O'Leary, D.J., Bentz, J.A., 1993. Ecosystem management model - Elk Island National Park: A case study of integrating environmental models with GIS. Proceedings of the 2nd Int. Conf. Integrating GIS and Environ. Mod., National Center for Geographic Information Analysis, University of California, Santa Barbara, CA, USA.

Campbell D.J., H. Gichohi, A. Mwangi, and L. Chege, 2000. Land use conflicts in S.E. Kajiado District, Kenya. *Land Use Policy* 17: 339-348.

Campbell, D.J., H. Gichohi, R. Reid, A. Mwangi, L. Chege and T. Sawin, 2002. Competition and conflict between people and wildlife in S.E. Kajiado District, Kenya. Working Paper Number 18 Nairobi: ILRI (www.lucideastafrica.org).

Campbell D. J., D.P. Lusch, T. Smucker, and E. E. Wangui, 2003. Land use change patterns and root cause in the Loitokitok Area, Kajiado district, Kenya. Working Paper Number 19, Nairobi: ILRI (http://www.lucideastafrica.org).

Christensen, L., M. B. Coughenour, J. E. Ellis, and Z. Z. Chen. 2004. Vulnerability of the Asian typical steppe to grazing and climate change. Climatic Change 63:351-368.

Coughenour, M.B., 1985. Graminoid responses to grazing by large herbivores: Adaptations, expectations and interacting processes. *Ann. MO Bot. Gard*: 72, 852-863.

Coughenour, M.B. 1992. Spatial modelling and landscape characterization of an African pastoral ecosystem: A prototype model and its potential use for monitoring drought. In: McKenzie, D.H., Hyatt, D.E., McDonald, V.J. (Eds.), *Ecological Indicators*, Vol. 1. Elsevier, New York, USA, pp. 787-810.

Coughenour, M., R. Boone, P. Thornton, and R. Howe. 2002. Integrated modeling and assessment of changing pastoral ecosystems – experience in East Africa, and potential for application in Mongolia. In: *Fundamental issues affecting sustainability on the Mongolian steppe* (T. Chuluun and D. Ojima, eds.). International Institute for Study of Nomadic Civilization, Ulaanbaatar, Mongolia. pp.313-326.

De Leeuw, J., H. H. T. Prins, E. C. Njuguna, M. Y. Said, and R. A. de By. 1998. Interpretation of DRSRS animal counts (1977-1997) in the rangeland districts of Kenya. Enschede, The Netherlands: International Institute for Aerospace Survey and Earth Sciences, Enschede.

Ellis, J. and M.B. Coughenour 1998. The SAVANNA integrated modeling system. In: Drylands: sustainable use of rangelands in the twenty-first century. (V. Squires and A. Sidahmed, eds.) IFAD Technical Reports. Rome. P.97-106.

Hoare, R.E., and J.T. Du Toit, 1999. Coexistence between people and elephants in African savannas. *Conservation Biology* 13: 633-639.

Hemp, A., 2005. Climate change-driven forest fires marginalize the impact of ice cap wasting on Kilimanjaro, *Global Change Biology* 11: 1013-1023

Kinyamario, J. I. 1996. NPP Grassland: Nairobi, Kenya, 1984-1994. Oak Ridge National Laboratory, Distributed Active Archive Center. Oak Ridge, TN: Available at: http://www.daac.ornl.gov

Leslie, P.H. 1945. On the use of matrices in certain population mathematics. Biometrika 33:183-212.

Ludwig, J. A., M. B. Coughenour, A. C. Liedloff, and R. Dyer. 2001. Modelling the resilience of Australian savanna systems to grazing impacts. Environmental International 27:167-172.

Mburu, J., R. Birner, and M. Zeller, 2003. Relative importance and determinants of landowners' transaction costs in collaborative wildlife management Kenya: an empirical analysis. *Ecological Economics* 45: 59- 73.

Oesterheld, M., C. DiBella, and H. Kerdiles, 1998. Relation between NOAA-AVHRR satellite data and stocking rate of rangelands. *Ecological Applications* 8: 207-212.

Reid, R., L. Gachimbi, J. Worden, D. Wangui, S. Mathai, S. Mugatha, D. Cambell, J. Maitima, B. Butt, H. Gichohi, and E. Ogol, 2004. Linkages between changes in land use, biodiversity and land degradation in the Loitokitok Area of Kenya. Lucid Working Paper Series Number 49, Nairobi: ILRI (www.lucideastafrica.org).

Thornton, P.K., S.B. BurnSilver, R.B. Boone and K.A. Galvin, 2006. Modeling the impacts of group ranch subdivision on agro-pastoral households in Kajiado, Kenya. *Agricultural Systems* 87:331-356.

Thornton, P.K., R.H. Fawcett, K.A. Galvin, R.B. Boone, J.W. Hudson, and C.H. Vogel. 2004. Evaluating management options that use climate forecasts: modeling livestock production systems in the semi-arid zone of South Africa. *Climate Research* 26: 33-42.

Thornton, P., K. Galvin and R. Boone, 2003. An agro-pastoral household model for the rangelands of East Africa, *Agricultural Systems* 76: 601-622.

WWF, 2000. Elephants in the balance: Conserving Africa's elephants. www.panda.org/resources/publications/species/elephants/index.html