CONSERVING MUSK DEER IN THE WILD: A COMPARISON OF DIRECT PAYMENT AND COMMUNITY WILDLIFE MANAGEMENT STRATEGIES

by

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B.Sc., University of Ottawa 2005

RESEARCH PROJECT SUBMITTED IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREE OF

MASTER OF RESOURCE MANAGEMENT

In the
School of Resource and Environmental Management
Faculty of Environment

Report No. 515

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SIMON FRASER UNIVERSITY

Summer 2011

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ABSTRACT

Community wildlife management and payment for ecosystem services are two promising strategies to conserve wildlife in developing countries. This research project applies a numerical simulation approach to compare both strategies in terms of conservation and economic development outcomes, using musk deer in Nepal as a case study. The optimal policy for a donor, who wishes to induce greater conservation outcomes, depends largely on the resource conditions such as biological growth rates, stock densities, and capture technologies. Community wildlife management performs well when resource conditions are good (e.g., higher stock levels) and/or when the technology is efficient at capturing animals. On the other hand, PES has the potential to induce better conservation outcomes at the margin of profit maximization and to serve as a more appropriate policy when stock sizes are too low. There is also the potential for a mix of both strategies to serve as the optimal policy.

Keywords: community wildlife management; payment for ecosystem services; direct payments; musk deer; conservation

ACKNOWLEDGEMENTS

Numerous individuals have provided support towards the development of this research paper. First, I would like to thank my supervisory committee for all of their guidance and support. Dr. Duncan Knowler, my senior supervisor, provided me with a unique research topic, generously shared his data sets on household surveys from Phortse, and gave me foundational knowledge in the realm of ecological economics. Dr. Sean Cox provided me with the skill sets required to delve into the world of simulation modelling. Dr. Cox also provided detailed feedback in earlier proposals and drafts, which encouraged me to become a more critical writer. Dr. Michael Green generously shared his wealth of knowledge of the musk deer conservation problem from his own research and personal experiences.

Dr. Keshav Raj Kenal, who was at the time the Director General of the Department of Forests Ministry of Forests and Soil Conservation in Nepal, shared information regarding wildlife policies in Nepal. Volker Homes and Alexey Vaisman, of Trade Records Analysis of Flora and Fauna in Commerce (TRAFFIC), provided several reports and supplementary information related to musk deer. Achyut Aryal shared experiences of his research on musk deer in Nepal. Dr. Xiuxiang Meng provided copies of journal articles he published on musk deer farms in China. Yang Shan Lee and Yongshu Fan, both students from the School of Resource and Environmental Management (REM), helped me retrieve and translate several journal articles on musk deer written in mandarin. Rupanada Widanage, a friend and fellow student in the

Environment and Development Research Group of REM, generously offered to chair my 699 defence. My loving wife, Eryne Donahue, proofread earlier drafts, provided emotional support and endured the glow of my laptop late into the night over several years in our small studio apartment.

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1 INTRODUCTION

Although wildlife generates use and non-use benefits, 21-36% of all known terrestrial mammals are threatened with extinction mostly via anthropogenic causes (Pimm et al. 1995, McKinney 1998, Gaston 2005, Schipper et al. 2008). Legal and illegal harvesting are the largest threats to terrestrial mammals, second only to habitat loss and degradation (Schipper et al. 2008). The illegal international trade of animal parts for traditional Asian medicine accelerates the decline of numerous endangered wildlife species, including tigers, rhinoceros, bears, pangolins, and musk deer (Cheung 1995). Perhaps not surprisingly, the world's highest concentration of threatened terrestrial mammals occurs in Asia & South-East Asia (Schipper et al. 2008, McNeely et al. 2009). Furthermore, China, with its large population base and growing middle class, is anticipated to increase global demand for wildlife products in the future (Zhang et al. 2008, McNeely et al. 2009).

The musk deer (*Moschus* spp.) is a small ungulate found throughout far-eastern Russia and twelve Asian countries in forested and mountainous areas. Among the unique characteristics of musk deer, males possess elongated canine teeth for defence as opposed to antlers found in many other ungulate species. In addition, and of particular interest for this study, the male deer secretes musk to mark its territory and possibly attract females (Green 1985). Such musk is highly valuable substance and is sold predominantly as an ingredient in traditional Asian medicine and to a lesser extent in the perfume industry. Over the last half century, over-harvesting of musk deer has led to drastic declines in their numbers and all species of musk deer are now threatened with extinction (Schipper et al. 2008). Concern over declining musk deer populations led to international and national efforts to protect them starting in the 1970s.

Countries with natural populations of musk deer (range states) often adopt one of two policies to protect musk deer populations: (1) enforcing a complete trade-ban or (2) regulating trade through extractive-use. The Convention on International Trade for Endangered Species (CITES) enables both of these policies at an international level through the listing of species under two Appendices. Appendix 1 prohibits commercial trade of the species while Appendix 2 regulates commercial trade so that it is not detrimental to the species. CITES currently lists musk deer populations in Afghanistan, India, Myanmar, Nepal and Pakistan in Appendix 1, and lists all other populations, including those in China and Russia, in Appendix 2. China and Russia are unique because they have national approaches to manage musk deer for commercial production/harvest.

The establishment of protected area networks is a common landscape-level management approach that countries implement in order to conserve wildlife populations listed under Appendix 1 and 2 of CITES. The International Union for the Conservation of Nature (IUCN) categorizes protected areas largely based on the level of human interaction/impact. All categories of protected areas infer that wildlife would receive a certain degree of protection. Appendix 1 listed species benefit the most under IUCN category Ia: *a strict nature reserve*; while Appendix 2 listed species would often be managed under IUCN category VI: *protected area with sustainable use of natural resources*.

Poor socio-economic conditions in many range states obstruct the success of the current management framework (i.e., of protected areas and trade bans) to protect wild populations of musk deer. Unfortunately, many range states are also developing countries with a myriad of other social problems that compete for scarce human and financial resources (Cooney 2001). Consequently, these countries often lack sufficient funds to enforce bans and regulations. Complete bans in trade are particularly severe because the introduction of a ban essentially advertises the rarity of the species and often leads to increases in price on the black market, which increases poaching incentives (Courchamp et al. 2006). Based on the estimates of one researcher, the mean value of musk increased approximately 10-fold on the global market after CITES listed Himalayan populations on Appendix 1 (Green 1986). However, actual declines in musk deer population and the associated decrease in supply could have also contributed to the rise in price.

Even if local communities choose to refrain from poaching due to the presence of a ban, there is little incentive to protect musk deer from *non-local* poachers or to preserve musk deer habitat. The introduction of a trade ban potentially eliminates a significant source of legitimate local revenue. An incentive to protect musk deer populations from poaching diminishes as musk deer are no longer an economic asset. Furthermore, the opportunity cost of retaining their habitat increases as communities may turn towards alternative land uses such as livestock grazing and agricultural development. At this point, there is only a cost in protecting them (Cooney 2001).

Since the 1950's, musk deer farms have operated in China in an attempt to supply domestic demand (Yang et al. 2003). Proponents of musk deer farms commonly argue that these operations have the propensity to flood the market with musk and lower its price, thereby deterring poachers from harvesting wild populations (Yang et al. 2003). However, the productivity and profitability of Chinese musk farms is low due to difficulties in the domestication of musk deer and high input costs (Parry-Jones and Wu 2001, Yang et al. 2003, Meng et al. 2006). Although rearing techniques are improving, Chinese farms only supply 0.3 to 1.2% of domestic demand (Parry-Jones and Wu 2001). Thus, the remainder of musk continues to originate from illegal sources. Currently, farming has not proven to deter illegal poacher from harvesting wild populations of musk deer (Green et al. 2006). In addition, the benefits local communities receive from state run and private operations are uncertain and not well documented.

1.1 Local Incentives: A Way Forward for Protecting Wild Populations?

The Conferences of the Parties (CoP) for CITES relaxed restrictions on the trade of some CITES listed species after recognizing the need and opportunity for incentive-based conservation, especially in developing countries (Cooney 2001). Harvesting of ranched populations (CITES Resolution 3.15) and the introduction of quota systems (CITES Resolution 5.21) expands potential avenues for exporting countries to legally trade products from wild populations of CITES listed species on the international market. In 1992, the CoP passed a resolution formally acknowledging the need for economically poor countries to develop valid trade markets: “commercial trade may be beneficial to the conservation of species and ecosystems and/or the development of local people when carried out at levels that are not detrimental to the survival of species in question” (CITES 1992, Resolution 8.3).

Although no community-based wildlife management (CWM) project presently exists for musk deer, whereby the state decentralizes management rights to the local level, the CITES resolutions mentioned above provide mechanisms that could potentially empower local communities to manage wild populations. In particular, harvesting musk via a live-capture and release strategy is a potential means of conserving wild musk deer populations (Green 1986, Wood et al. 2008). Implementing a live-capture and release strategy, as opposed to a harvesting quota strategy, is less risky as the former maintains a musk deer population over time. Furthermore, local communities would have a greater incentive to protect musk deer from illegal poaching by owning clear management rights and benefiting from musk sales.

Governments or international donors could potentially finance or provide in-kind support to CWM projects. Subsidies could be offered when there are insufficient financial incentives or other barriers for enterprises to establish or expand on their own accord. At present, very little public information is available on financial subsidies for the extractive-use of CITES listed species. However, such subsidies are known to exist for crocodiles (*Crocodylus* spp.) in several developing countries (Hutton et al. 2001), Atlantic bluefin tuna (*Thunnus thynnus*) in the Mediterranean (De Stefano and Van Der Heijden 2007), and vicuña (*Vicugna vicugna*) in South America (Lichtenstein 2010).

Donors (i.e., either international, government or non-government organizations wishing to invest in conservation) have moved their financial support towards projects and programs that apply a community-based approach for wildlife conservation after international calls for participatory conservation in 1980s and 1990s (e.g. World Conservation Strategy 1980, Our Common Future (Brundtland Report) 1987, and

Convention on Biodiversity 1992). Community-based wildlife management, otherwise referred to as community-based conservation, is favoured among donors because it often alleges to improve both conservation and development outcomes for target communities (Campbell and Vainio-Mattila 2003). Successful CWM schemes occur when communities can engage in significant wildlife management decisions and receive benefits from conservation (Gibson and Marks 1995, Mehta and Heinen 2001). Local economic development opportunities are often a component of CWM schemes (Gibson and Marks 1995, Mehta and Heinen 2001). For instance, in Tanzania, donors subsidized income-generating activities in order to win local support for conservation programs (Songorwa 1999). CWM is now a “major narrative” and a preferred strategy among multi-lateral, bilateral and non-government funding agencies involved in conservation and development (Campbell and Vainio-Mattila 2003).

Evaluating the impact and effectiveness of CWM projects/programs is prudent given the emergence of CWM as a dominant conservation approach and the large amounts of funds flowing towards these projects (Campbell and Vainio-Mattila 2003). The needs and priorities of local communities are often at odds with the conservation objectives (Brandon and Wells 1992). Some critics argue that biological conservation issues are often set aside in order to address human development goals (Ferraro and Simpson 2002). For instance, local-level forest policy developed by communities in Nepal resulted in insufficient guidelines to conserve biodiversity within their local statutes and operational plans for forest resources (Khadka and Schmidt-Vogt 2008). The plans placed more weight on economic (e.g., firewood, herbal medicine, and income generating activities) and general conservation objectives (e.g., grazing control and soil erosion) (Khadka and Schmidt-Vogt 2008). As another example, benefits received from

CWM programs, operating on the continent of Africa, were insufficient to deter households from illegally poaching (Gibson and Marks 1995, Songorwa 1999).

Sceptics question the effectiveness of community based conservation not only for its disputable conservation outcomes but also for its poor economic and development outcomes {{97 Brandon,K.E. 1992; 367 Songorwa, A. N. 2000; 300 Ferraro,P.J. 2001}}. In a review of 37 subsidized community-based conservation enterprises, only seven were profitable at the time of the study and less than half were able to cover their variable costs (Salafsky et al. 2001). Devolution of power over natural resources to communities can lead to an escalation in the divide between local elites and the poor. Wealthy stakeholders are prone to have greater influence on community-level decisions and, unless regulated, are able to make decisions to satisfy their own interests (Gibson and Marks 1995, Hughes 2001, Campbell and Vainio-Mattila 2003, Mansuri and Rao 2004). Given the challenges facing CWM, two separate programs may be more cost-effective than trying to implement a single one to tackle both conservation and development problems (Simpson and Sedjo 1996). Focusing on these issues separately removes constraints on the project/program and allows for the targeting of a specific objective.

Of the financial mechanisms available, a more direct approach that has received quite a bit of attention in the last decade is payment for environmental services (PES) (Gomez-Baggethun et al. 2010). PES schemes are broadly defined as initiatives having a voluntary buyer, or group of buyers, who conditionally purchase an environmental service from a voluntary supplier or group of suppliers (Wunder 2006). Under a PES scheme the supplier retains clear property rights over the particular land or resource that produces the environmental service, and the supplier is influenced by market incentives to provide the service in sufficient quantities. Support for PES schemes are often a response to inefficiencies in command-and-control regulations because PES provides benefits directly to communities (Engel et al. 2008, Jack et al. 2008). In addition, an important distinction from other financial incentives is that the funds are conditional on the provision of the service. Under a broad array of scenarios, the removal of payments to additional lines of production (i.e., removal of payments that indirectly attempt to encourage the provision of environmental services by supporting "eco-producers" via capital subsidies or price premiums) leads to a gain of efficiency (Ferraro and Simpson 2002, Wunder et al. 2008).

Given the emergence of CWM and PES as potential conservation policies for conserving musk deer, an evaluation needs to take place not only surrounding their ability to conserve wildlife species but also on their potential to address human development needs and on their cost-effectiveness for donors.

1.2 Research Objectives

The objective of my research is to compare a CWM to a PES strategy using musk deer in Nepal as a case study. Both strategies potentially provide new opportunities to conserve wild musk deer populations and both target local communities. I will evaluate the strategies using a numerical optimization approach that takes into account biological and economic information relevant to musk deer. I explore some of the main conservation and development outcomes of the two strategies in order to compare the relative suitability of PES and CWM schemes.

1.3 Scope and Limitations

Since no such PES or CWM scheme currently exist for comparison, this research develops a hypothetical CWM and PES scheme in order to compare the conservation and development outcomes for musk deer. More specifically, with respect to conservation outcomes, I compare the cost effectiveness of either scheme in terms of the steady state equilibrium that musk deer population approach over the long term. In terms of development outcomes I compare the benefits communities receive under either strategy.

CWM and PES strategies can take on a multitude of potential designs for wildlife management, and there is no doubt that improvements to the designs and modelling approach proposed in this research paper could help address context specific situations, and take advantage of new knowledge gained from additional studies. For instance, only the main operational cost of either strategy over time that a hypothetical donor would pay is incorporated (i.e., not set-up or transaction costs). In the PES scenario, transaction costs may include monitoring costs to ensure that the service provider upholds their contract to provide a conservation outcome. Monitoring costs, such as a monitoring program to assess the trend in musk deer populations, could be prohibitively expensive for PES to operate. Incidentally, similar monitoring costs may not apply to a CWM scheme. Furthermore, I apply a numerical optimization model in this paper to take advantage of some of the available information on musk deer. Although a general analytical model would provide additional insight into the relative suitability of either strategy, such a model is beyond the scope of this project.

2 LITERATURE REVIEW

In this chapter, I first explore the current management framework for musk deer in Nepal and the potential for community wildlife management via a live-capture and release strategy. Then I review current payment for ecosystem services strategies for managing terrestrial wildlife. Next, I review the literature on the potential advantages and disadvantages of PES in comparison to CWM with respect to conservation and development outcomes. Finally, I summarize some of the existing literature that compares conservation policies for terrestrial wildlife in order to provide a framework for comparing CWM to PES strategies for musk deer.

2.1 Musk Deer in Nepal: Case Study and Potential for CWM

The IUCN classifies all three species of musk deer (i.e., *M.chrysogaster, M.fuscus, M. leucogaster*) in Nepal as endangered (Timmins and Duckworth 2008, Wang and Harris 2008a, Wang and Harris 2008b). For this reason, all musk deer populations in Nepal are included in Appendix 1 of CITES, which effectively bans the international commercial trade of these species or products made from their derivatives. The government of Nepal formally supports the international ban through the National Parks and Wildlife Conservation Act (NPWCA 1973), which prohibits the killing and injuring of musk deer and trading of musk glands within the country. The rights to manage and protect wildlife remain in control of the Nepalese government where park and military personnel enforce wildlife laws (Bhudhathoki 2003).

Despite a legal structure for protection, musk deer populations in Nepal are still threatened by illegal poaching activities. In the mid 1980s, poaching activities annually removed an estimated 15% to 53% percent of the Himalayan population of musk deer, which may have reduced the entire population to 10% of natural levels that the landscape could support (Green 1986). Although a more recent estimate of the status of the Himalayan musk deer population is not available, poaching continually threatens musk deer populations in this region (Subba 2000, Aryal et al. 2010).

The Nepalese government faces a number of challenges with respect to musk deer conservation, including geographical and political constraints. Part of the difficulties in enforcing the law is that musk deer occupy steep terrain in some of the most remote areas of the country. Few reliable transportation networks make enforcement costly. Until recently, Nepal's unstable political climate further jeopardized long-term conservation goals. Within the last decade, Maoist insurgents seized several national park guard posts (Baral and Heinen 2006). Although the insurgents' interests were mainly political, and occupants were not necessarily harming wildlife directly, the take-over of these posts led to additional uncertainty of park enforcers' ability to prevent poaching activities (Stubblefield and Shrestha 2007). This highlights the need for more local involvement given that rural communities have a closer and more permanent relationship with the land than a military or park guard (Bhudhathoki 2003).

Recognizing that Nepal is a developing country and is resource poor, a potential solution for conserving both wild populations of musk deer is to promote a live-capture and release harvest scheme (Green 1986, Green 1989, Wood et al. 2008). Under this scenario, animals could be live-captured from the wild where their musk would be ‘milked’ and then released to be captured again in the future (Kattel and Alldredge 1991). The community could sell the harvested musk on the international market. If Nepal and the international community permitted a CWM project through changes in national and international law, then participating communities may have an economic incentive to conserve musk deer populations in order to ensure a profitable stock over the long-term. Indeed, a live-capture and release scheme for harvesting wool fibers from vicuña, a llama-type species, is partly associated with the successful population recovery of vicuña in many parts of South America (Lichtenstein 2010).

A cost-benefit analysis (CBA) of a hypothetical live-capture and release project for wild musk deer suggests that positive economic returns are possible (Wood et al. 2008). Much of the data for the CBA comes from Phortse, a small village within Mt. Everest (Sagarmatha) National Park. However, the transferability of these hopeful economic findings to other areas of Nepal is questionable for at least two reasons. First, a large military outpost in the park likely contributes to a lower frequency of visits by poachers in comparison to other areas. It is important to note, the CBA did not consider enforcement costs (Wood et al. 2008). Secondly, the supposed population density of musk deer was roughly 10 times higher than average population densities of un-poached musk deer assumed to occur in other areas of the Himalayas.[1] The low densities and cryptic nature of many musk deer populations negatively influences the economic

[1] Green (1986) estimated that un-poached populations of musk deer in the Himalayas could conservatively reach 4-6 individuals/km^2 based on a study in Kedarnath National Park, India. However, the nature park experienced poaching activities prior to his arrival and may contribute to a rather conservative estimate. High carrying capacity estimates in Phortse of up to 46 individuals/km^2 is likely due to decades of un-poached populations, but may also reflect exceptionally high quality habitat (e.g., the availability of high quality food resources for musk deer or favourable climatic conditions).

potential of live-capture and release enterprises (Harris and Guiquan 1993, Parry-Jones and Wu 2001).

2.2 Potential for PES in wildlife conservation, and the pros and cons

Another option is to pay local communities directly for the conservation of musk deer through a payment for ecosystems services (PES) scheme. So far, the application of PES for conserving wildlife populations has largely focused on the protection of large carnivores. These schemes typically identify livestock herders and farmers as the service provider since these individuals incur damages from predators to livestock and crops. In Sweden, the government pays Sami villages directly for the number of certified newborn predators (i.e., wolverines, wolves and lynx) on their rangelands as opposed to providing compensation for livestock damages (Zabel and Holm-Mueller 2008). The level of payment for each offspring is equivalent to the anticipated loss in revenue the offspring is expected to cause during its lifetime (Zabel and Holm-Mueller 2008). Live-stock ranchers in Sonora, Mexico are paid for each photograph that “camera-traps” take of jaguars on their property (Nelson 2009). In India, the village of Kibber is paid to retain a 500 ha “no grazing zone” to protect habitat for bharal, which are natural prey for snow leopards (Nelson 2009).

Several key papers explore the efficiency of PES to other forms of subsidies that attempt to increase the provision of an environmental service. Ferraro and Simpson (2002) use an example of a donor who has the choice between subsidizing the cost of forested-land (direct subsidy) versus subsidizing some other input (indirect subsidy) for operating an eco-entrepreneur business (i.e., a profit maximizing business that has an environmental service as one of its inputs). Ferraro and Simpson (2002) show that buyers of environmental services will prefer direct subsidies to indirect subsidies when own-price effects are stronger than cross-price effects on the margin of profit maximization. Specifically, an environmental service is provided by the eco-entrepreneur at a greater quality/quantity through a reduction in price of that service, rather than a reduction in the price of some of other input. Under the assumption of perfect markets and when eco-entrepreneurs are operating at profit maximizing levels prior to a subsidy, the buyer will prefer a direct subsidy for any homothetic technologies (Ferraro and Simpson 2002). In the context of the production function, homothetic technologies are when "expansion paths are rays, i.e., the optimal relative [input] factor mix remains the same for all levels of output" (Lindberg et al. 2002). In contrast, the local recipient of the funds will prefer the indirect strategy because they receive more funds in order to reach a desired quality/quantity of an environmental service (Ferraro and Simpson 2002). A dynamic analytic study (i.e., a study that follows state dynamics over a time horizon as opposed to a single "static" period), using similar market assumptions, concluded that the cost-minimizing environmental service buyer nearly always prefers direct payments, followed by eco-premium payments (i.e., a premium on the sale price of an ecofriendly product) and then by capital subsidies as a last resort (Ferraro et al. 2005).

Although the theoretical analysis of Ferraro and Simpson (2002) is often cited in support of the economic efficiency of PES (Wunder 2007, Engel et al. 2008, Groom and Palmer 2010), only a few studies subsequently explore alternatives to the underlying assumption of perfect markets. Perfect markets, after all, are the exception rather than the norm in many developing economies (Muller and Albers 2004, Groom and Palmer 2010). Under some imperfect market conditions, a preference for indirect subsidies can prevail for both the donor and recipient when subsidies enable recipients to overcome major financial barriers due to lack of access to credit, and when eco-production technologies link closely to the conservation needs of the land (Groom and Palmer 2010). Furthermore, under scenarios of missing labour or resource markets, the optimal policy can differ among a mix of agricultural projects, direct payments, and enforcement interventions since these market conditions influence a household's response to various interventions (Muller and Albers 2004).

Since Ferraro and Simpson (2002) focus on marginal analysis, they imply that the eco-entrepreneur is *currently* in business and maximizing profit prior to any donor intervention. Therefore, the international donor only wishes to induce greater conservation outcomes than the eco-entrepreneur would otherwise engage in. However, this is not the case in Nepal, as no CWM project for musk deer currently exists. If the recommendations of Ferraro and Simpson are taken at face value, then a less critical and impulsive donor may focus on a PES strategy and neglect the conservation potential of a CWM project. From this perspective, the complete abandonment of a CWM could lead to lost conservation opportunities.

There are other criticisms of PES beyond arguments of cost-effectiveness. A common concern is that these payment schemes often require sustained long-term funding because they do not invest in eco-entrepreneur businesses (Swart 2003). Secondly, PES has the potential to erode local social and traditional values associated with the natural resource by putting a price on them (Swart 2003). Another concern among potential buyers of environmental services, who often have multiple objectives (i.e., conservation and development goals), is that they will often hesitate to finance a policy that focuses solely on addressing a conservation objective, especially in a developing country (Wunder et al. 2008). For these reasons, finding prospective donors that are willing to sustain PES programs into the indefinite future is difficult.

Conversely, proponents of PES suggest that numerous subsidies to eco-entrepreneurs also require continuous funding (Wunder et al. 2008). Proponents of PES argue that if the enterprise in question is not profitable, then funds would be better spent directly on conservation (Ferraro and Simpson 2005). With respect to concerns surrounding the erosion of social or traditional values, PES supporters suggest that most schemes operate in locations where there is an apparent risk to the environmental service. Thus, some form of intervention is necessary because the social/traditional values that upheld the environmental service(s) in the past are unlikely to sustain the service(s) into the future (Simpson and Sedjo 1996, Ferraro and Kiss 2003, Wunder et al. 2008). Finally, in response to concerns of a donor with a dual objective of conservation and development, some studies show that under particular circumstances, such as unique resource conditions or imperfect markets, that there are potential "win-win" opportunities with the PES approach. For example, paying Maasai not to grow fenced-in crops, in order to maintain elephant foraging habitat, has the potential to increase the local community's welfare because pay-off from a PES scheme is potentially greater than the payoff from growing fenced-in crops (Bulte et al. 2008, Groom and Palmer 2010). In addition, even if the recipient prefers an indirect subsidy (i.e., provided that they receive more funding from an indirect subsidy), it is also possible that a transfer (i.e., side-payment) to the recipient of some of the cost savings the buyer receives from applying a PES scheme could leave the recipient better off (Ferraro and Simpson 2002)

Although PES is not a panacea, this financial mechanism provides a very direct and promising link to address the conservation of wildlife populations. On the other hand, subsidizing a CWM project that involves the live-capture and release of an endangered species is arguably one of the better-linked conservation approaches within the "community-incentive" portfolio as it promotes the management of the species as opposed to the production of some other asset.

2.3 Models for Comparing Wildlife Policies

The literature comparing incentive-based wildlife conservation policies is slowly expanding. The following section reviews some of the various wildlife models used to compare policies in developing countries.

Skonhoft and Stolstad (1998a) develop a basic model with two main agents: a resource owner and a local community whom has no legal rights to manage wildlife. The resource owner can optimize their utility by allocating effort between harvesting wildlife and anti-poaching activities. The resource owner also receives benefits from non-consumptive use of wildlife (e.g. tourism revenue), which increases with a greater wildlife stock. Meanwhile, the local community optimizes their household utility by dividing their time between agricultural production and illegal harvesting. In this scenario, the land-base is fixed so that wildlife and agricultural land is separate. By excluding the possibility of land conversion, the dynamics of the model focuses on the impact of legal and illegal harvesting on a single wildlife stock. Under such a scenario, the use of an agricultural subsidy helps to reduce illegal poaching efforts because the community shifts their effort to agricultural production. However, a subsidy provided to the resource owner to offset the cost of anti-poaching effort is ambiguous. If the community does not respond to an increase in enforcement effort, subsidizing anti-poaching effort is ineffective. On the other hand, if the community dominates the off-take of wildlife through illegal poaching and is even slightly deterred by an increase in anti-poaching effort, then subsidizing anti-poaching effort will unambiguously increase the wildlife stock (Skonhoft and Solstad 1998a).

Although wildlife is not explicitly identified in their model, Muller and Albers (2004) explore a similar scenario to Skonhoft and Stolstad (1998a). Muller and Albers (2004) compare how a local community adjusts their level of resource extraction (e.g. firewood extraction) from a protected area when confronting a mix of policy interventions, including: (1) an agricultural subsidy, (2) a conservation payment not to extract (i.e., a type of PES strategy), and (3) an increase in enforcement against resource extraction. The authors highlight how missing labour markets (i.e., the lack of a free flow of labourers willing to provide their service for a fee), missing resource markets (i.e., lack of tradable commodities in the market place) or both can affect the optimal mix of policies. For instance, when a resource market is lacking then the enforcement and the PES strategy is less effective, in comparison to a non-missing market scenario (i.e., perfect market), as communities still need to extract a minimum amount of resources to satisfy their household needs. If the labour market is missing, then again enforcement and PES strategies are less effective. Interestingly, Muller and Albers (2004) research suggests that the Skonhoft and Stolstad (1998a) model implicitly assumed a missing labour market. If a labour market existed then the effects of an agricultural subsidy would have no effect on resource extraction, as the community could buy labour from outside of the community. However, missing markets are quite common in remote areas of developing countries where conservation interventions are needed (Muller and Albers 2004).

Fisher, Muchapondwa and Sterner (2005) explore a slightly different scenario where the local community does not directly engage in illegal poaching but they can either encourage outsiders to poach or the community can engage in anti-poaching effort themselves. In this model, the authors show that benefit sharing, of legal hunting revenue or tourism revenue, from a resource owner, such as a park manager, can encourage more anti-poaching effort by the community but that such a response is not fail-safe. If the resource owner shares benefits of legal hunting revenue, then the community will only increase their anti-poaching activities if such effort leads to more hunting-licenses. If benefits are provided as a share of tourism revenue, then the community increases anti-poaching effort only if the wildlife stock is allowed to increase (i.e., the stock is not completely offset from an increase in hunting quota) (Fischer et al. 2005).

In another model, Skonhoft and Stolstad (1998b) explore the possibility of a local community that engages in both wildlife harvesting and livestock herding on the same land-base. Under this scenario, livestock and wildlife populations compete for the same resources. The community also enjoys un-restricted access to the wildlife resource. Thus, in comparison to Skonhoft and Stolstad (1998a), a subsidy to increase livestock prices can reduce wildlife population levels because of an increase in the opportunity-cost for conserving wildlife. On the other hand, the use of a subsidy to increase the value of the wildlife unambiguously increases the wildlife stock. This latter subsidy can include either an increase in the revenues received from wildlife off-take sales or an international transfer payment subject to the wildlife stock level (i.e., a type of PES strategy). The efficacy of this subsidy relies on the community retaining management rights over the wildlife stock so that they have a long-term interest in sustaining it (Skonhoft and Solstad 1998b).

Much of the literature on wildlife policy options, including the four papers discussed above, apply an analytical bio-economic model to explore marginal responses in wildlife to a policy intervention at a steady-state equilibrium. The benefit of this approach is that it keeps results quite general and tractable. Although they provide many insights into the qualitative response in wildlife stock to a policy intervention (positive or negative growth), they fall short in comparing the quantitative differences of policies that are deemed effective. This gap is mostly due to the paucity of data and knowledge of appropriate functional forms.

Zabel and Holm-Muller (2009) are the first researchers to compare the cost-effectiveness of PES to another incentive-based wildlife conservation strategy (i.e., compensation payments). The authors first develop an analytical model with a group of livestock herders that optimize their utility by dividing effort between off-farm labour and killing carnivores. In this model, the livestock herders cull carnivores to reduce wildlife damages to their livestock asset, but do not receive additional benefits such as sale from game-meat or animal parts. The authors fit their analytical model to available data for tigers in India, which allows them to make some quantitative comparisons. Their analysis shows that the total costs of either PES or compensation payment intervention is contingent on the predator-prey functional response. If the number of livestock killed per predator is higher at the socially optimal wildlife equilibrium level than the average number killed per predator, then PES is more cost-effective. If lower, then compensation payments are more cost-effective. Assuming a linear functional response, the authors suggest that costs using either strategy are equal (Zabel et al. 2009).

In summary, among the conservation approaches available, CWM and PES schemes offer promising avenues for musk deer conservation. The cost benefit analysis from Phortse Nepal, suggests that CWM is economically viable under favourable resource conditions, but lacks an incorporation of poaching dynamics and poorer resource conditions that are more representative of the majority of locations that musk deer inhabit. Conversely, PES could provide a more cost-effective approach based on theoretical analyses. However, missing markets and lack of access to credit, which are common in developing countries, are often not included in theoretical models. In addition, most analysis promoting PES compares strategies at the margin but does not consider conservation outcomes from the entire project. Although comparisons of wildlife conservation schemes are expanding in the literature, no literature compares a CWM project to PES in any detail.

3 METHODS

The first section of this chapter introduces the framework developed to compare a community-based wildlife management scheme to payment for ecosystem services scheme. The second section describes the study area and data sources, which inform the resource problem and help to identify baseline parameters respectively.

3.1 Modelling Approach

The model I develop in this paper takes some of the components described in the available literature, but also adds a few unique assumptions. My research helps to address some shortfalls in the original cost benefit analysis by Wood et al. (2008) by introducing poaching and considering search effort in more detail. Similar to Skonhoft and Stolstad (1998a) I focus on pressures from poaching, rather than agricultural expansion, so land conversion is not considered in this model. Also following Skonhoft and Stolstad (1998a) as well as Zabel and Holm-Muller (2009), I assume the labour market is missing, so there is a constraint on the community's time endowment. The assumption of a missing labour market appears consistent with many remote communities in developing countries (Muller and Albers 2004). The assumption of a missing resource market is not an important component, since musk is poached mostly for trade on an international market as opposed to local consumption. As in the model by Fisher, Muchapondwa and Sterner (2005), I assume that pressure from poaching originates from outside the local community because musk deer poachers are frequently cited as "outsiders" (Stubblefield and Shrestha 2007). This is often the case for a number of Tibetan Buddhist communities that live next to Himalayan musk deer populations where injuring animals is forbidden within their culture (Harris 1991, Stevens 1997, Mishra et al. 2006). By contrast, the community may engage in the legal harvest of musk via the live-capture and release of musk deer, which is assumed to have negligible impact on the wildlife stock. Forest and wildlife resources in developing countries are often under a *de-facto* open-access regime (Bluffstone 1995). Enforcement to secure resources can remain a significant component of resource management when resource ownership is poorly acknowledged (Kuperan et al. 1999). Since poaching is the largest threat to musk deer in Nepal, financing community-based enforcement may serve as an appropriate PES type scheme. Although, the removal of an ecosystem threat is not common among PES schemes, a similar type of PES contract was suggested for invasive species removal in Africa in order to improve watershed services and benefits for biodiversity (Turpie et al. 2008). The environmental service under the musk deer scenario would be the improvement of biodiversity benefits by securing a population of musk deer via the partial or full removal of a poaching threat. Applying this assumption to a PES scheme means that the community should be paid to engage in more anti-poaching effort than they would normally engage in without an intervention (i.e., they are paid to enforce, and are not paid to refrain from poaching as in Muller and Albers (2004)). Given the level of benefits they receive from either a community-based wildlife harvesting scheme or a payment for ecosystem services scheme, the local community chooses the level of anti-poaching effort to engage in.

The model framework considers four main agents: (1) a hypothetical community living near a population of musk deer, (2) outside poachers, (3) international donors, and

(4) musk deer.[2] Poachers originate from outside the community and threaten musk deer population viability by engaging in unsustainable poaching activities on an annual basis. The main objective of the international donor is to identify the most cost effective means to maintain a musk deer population at a target stock level. The community does not participate in anti-poaching activities without some sort of incentive, so the donor will prefer the option that increases the community's anti-poaching efforts in order to maintain the musk deer population. However, the donor also wants to support a policy that contributes to the overall economic-development of the community. Therefore, both biological and economic outcomes are compared between the two strategies. The international donor has the following two options for conserving musk deer: (1) indirectly support anti-poaching effort by contributing funds to a budget that attracts participants to engage in a CWM project, or (2) pay community members directly to engage in anti-poaching activities through a PES scheme (Figure 1). In the CWM option, an additional stream of revenue is available due to the legal sale of musk. In this case, the community must optimally allocate their effort between anti-poaching and harvesting in order to maximize the returns per individual community member per year. Under the PES scenario, every dollar spent by the international donor contributes only to enforcement. The annual returns/payments are spread evenly among participating individuals. I conduct a numerical optimization analysis in order to compare CWM to PES strategies

[2] If the musk deer population is within a protected area, then there may also be a fifth player: a protected area (PA) manager who is ultimately the responsible authority recognized by the government for ensuring the protection of musk deer. Although this research paper does not explore this scenario, the PA manager may be involved in either PES or CWM strategies in several ways. For example, the PA manager may serve as a contract service "broker" between the local community and the donor. In this case, the PA manager could take the role of a third party evaluator to ensure the environmental service was provided by the community. On the other-hand, the implementation of either a PES or CWM strategy, when the PA manager is already investing in enforcement, may change how the PA manager optimizes their own enforcement effort.

for conserving musk deer. Code for the model was developed using the open-access software package *R* (R Development Core Team 2010).

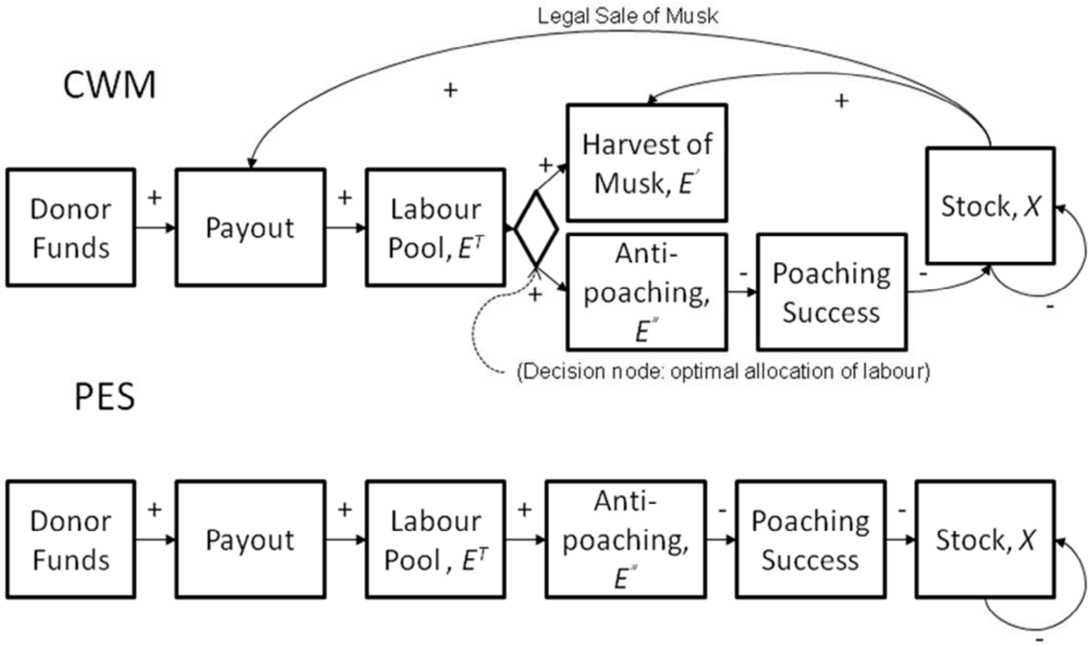

Figure 1. Modelling schematic for comparing a community-based wildlife management scheme to a payment for ecosystem services scheme. PES scheme is simply a restricted version of the general CWM framework.

3.1.1 Community Wildlife Management Model

When the model community engages in harvesting via live-capture and release of musk deer, households in the community are assumed to maximize their utility over time based on the following equation:

$$\max_{E',E''} U = \sum_{t=1}^{\infty} \pi_t \rho^t = \sum_{t=1}^{\infty} \left[\frac{pH_t(E_t', X_t) - (c_t - z)}{I_t(\pi_t)} \right] \rho^t \qquad [1.1]$$

where U is utility, t is a time index incremented by year, p is the price per musk harvest from one deer, H_t is the number of musk deer harvested in year t, c_t is a fixed cost, c (i.e., cost of medicine, propane and nets) if harvesting occurs in year t (i.e., $c_t = \begin{cases} c & H_t > 0 \\ 0 & H_t = 0 \end{cases}$), z is a fixed annual subsidy, I_t is the number of individuals participating in CWM in year t, π_t is profit per individual, and ρ is the discount factor. By design, utility is the maximization of individual profits over the time horizon. The harvest of musk is dependent on the total person hours allocated to harvesting $E_t^{'}$, as well as the population size X_t. Since the community cannot extract musk from the same deer within a given year and due to the presence of handling time, the dynamics of harvesting are non-linear (described in Section 4.3). Equation 1.1 is also subject to the following constraints:

$$X_{t+1} - X_t = F(X_t) - P(X_t, E_t^{''}, I_t) \quad [1.2]$$

$$E_t^T = E_t^{'} + E_t^{''} \leq \overline{E} \quad [1.3]$$

$$I_t(\pi_t) = \Phi\,(\Gamma_t)Y \quad [1.4]$$

$$E_t^T = I_t l \quad [1.5]$$

$$E_t^{'} = 0 \text{ whenever } I_t < Y^m \quad [1.6]$$

$$E_t^{''} = E_t^T \text{ whenever } I_t < Y^m \quad [1.7]$$

Equation [1.2], the equation of motion, describes the population dynamics. The function F is a natural growth function for musk deer and the function P is the amount of off-take from illegal poaching, which itself is dependent on the size of the stock, the total person hours allocated to enforcement effort, $E_t^{''}$, and the number of individuals participating , I_t, in year t. Anti-poaching dynamics are also non-linear due to handling time (See section 4.4.2 for more detail).

Equation [1.3] describes the constraint on the labour market. E^T is the total number of hours allocated to CWM in a particular year and $\overline{E}$ is the total number of hours available from the community (i.e., a labour constraint).

Equation [1.4] describes the number of individuals that participate in CWM in a given year. The number of individuals willing to participate is dependent on anticipated profit per individual, π_t, which depends on the sale of musk and the donor subsidy received. The use of the term $\Phi\ (\Gamma_t)$, on the right-hand-side, is unique to wildlife conservation models and a full explanation of the term is provided in Section 4.2.

Equation [1.5] shows the scalar relationship between individuals willing to participate and the total number of hours committed to the CWM project. The l parameter is the total number of hours provided per individual per year.

Equation [1.6] and [1.7] accommodates a minimum constraint on individuals required for harvesting musk deer. I assume the community applies a drive-net technique as their preferred live-capture and release strategy for harvesting musk. This technique requires 10 to 15 individuals for the successful capture of musk deer (Kattel and Alldredge 1991). If the total number of individuals participating, I_t, in any year is less than the minimum number of individuals for harvesting required, Y^m, then harvesting effort, $E_t^{'}$, cannot be applied and enforcement effort, $E_t^{''}$, remains the only viable activity.

3.1.2 Dynamic Programming for Optimal Allocation and Division of Labour

Since non-linear dynamics are inherent in the CWM model due to capture and enforcement dynamics (refer to sections 4.3. and 4.4 below), and due to the presence of fixed costs (see also Rondeau and Conrad 2003), the solution to equation [1.1] cannot be obtained analytically. Instead, I solve the problem recursively using the dynamic programming equation:

$$V(t,X_t) = \max_{E_t',E_t''} \left\{ \rho^t \left[pH(E_t', X_t) - (c - z) \right] / I_t(\pi_t) \right\} + \rho^{t+1} V(t+1, X_{t+1}) \qquad [3]$$

where ρ is the discount factor, $V(t,X_t)$ is the present value function or the maximum net present value in period t given the stock X_t with the allocation of harvest, E_t', and enforcement, E_t'', and assuming that the remainder of the management program in future years is optimal. The value at the terminal time, T+1, pushed far enough into the future, is zero since the community discounts their future revenue streams. Assuming no value in the future stock, the second term on the right hand side is set to zero in the last management period. The optimal controls, $E'^{,*}$ and $E''^{,*}$, are determined for the entire possible range of stock levels from the terminal time period to the initial period so that an optimal control policy can solve for any initial stock level.

3.1.3 Payment for Ecosystem Services Model

In contrast to the community harvesting approach, the payment for ecosystem services scheme is much simpler. Since the harvesting option is not considered, the annual subsidy provided by the donor is the only applicable stream of revenue entering the community. The amount of enforcement effort, E'', and the number of individuals participating is directly dependent on the subsidy received.

3.1.4 Comparison of PES to CWM

Conservation outcomes and the relative cost effectiveness of either the PES or the CWM strategy is determined by comparing the equilibrium of musk deer population levels over a similar time-horizon given the same annual subsidy. Musk deer stocks are set at 20% of carrying capacity, K, and run under CWM and PES scenario for 80 years (refer to Section 4 for full model description). Many of the results in Section 5 display how conservation and economic outcomes differ between CWM and PES strategies when the annual subsidy, z, is calibrated to obtain equilibria of 50, 70 and 90% of K under the PES scenario. Furthermore, the net present value (NPV) of economic benefits accruing to the community is used to compare development outcomes.

3.2 Study Area Description and Data Sources

The model developed to compare CWM to PES strategies was largely based on several studies conducted in Sagarmartha (Mt. Everest) National Park, part of the Khumbu Region of Nepal (See map) (Kattel and Alldredge 1991, Kattel 1992, Knowler et al. 2004, Wood et al. 2008, Aryal et al. 2010). In particular, a number of studies were based out of Phortse, a small Tibeto-Buddhist community. Musk deer reside in the park at altitudes between 3400 m and 3900 m above sea level (Aryal et al. 2010). The sub-alpine forests in these regions provide habitat for a relatively dense population of musk deer. The community of Phortse depends on subsistence agriculture and on employment or business opportunities from a steadfast and growing tourist industry. Largely due to favourable economic opportunities and religious beliefs, Phortse has little incentive to poach musk deer. In comparison to other regions in Nepal, Phortse is unique given the relatively high density of musk deer and its healthy economy. However, as described below, a sensitivity analysis will explore the less ideal conditions experienced in other areas of Nepal.

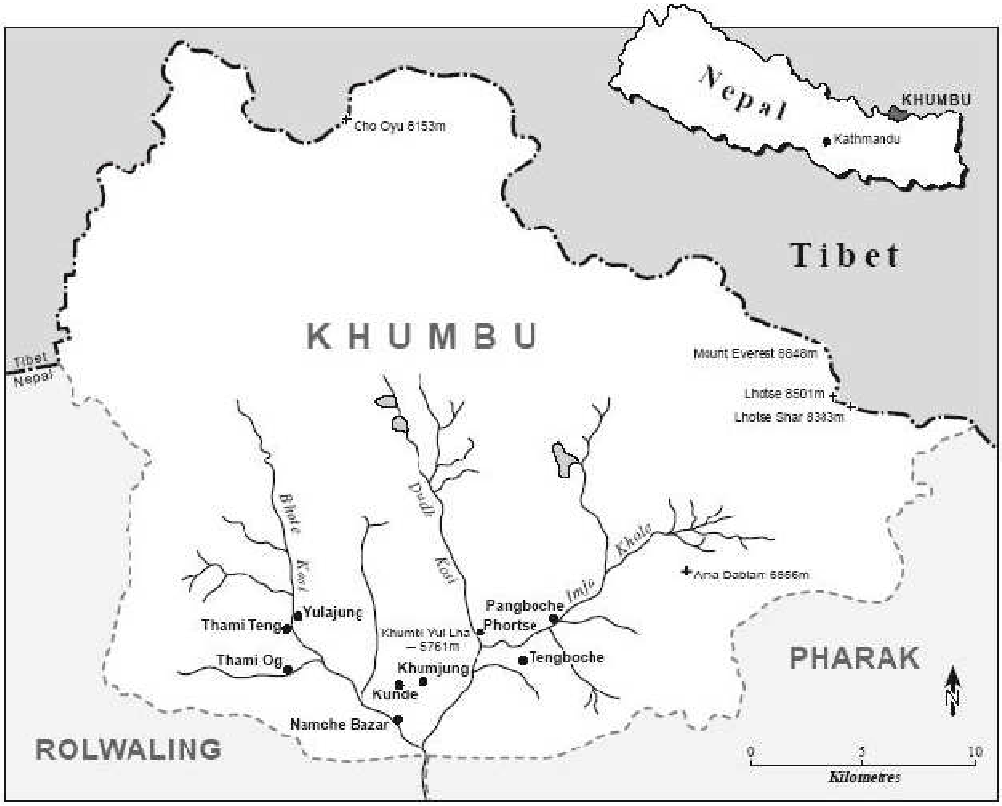

Figure 2. Khumbu region of Nepal. Map source: (Knowler et al. 2004)

In the early 1990s, a research team based out of Phortse monitored and captured a number of musk deer to study their ecological and biological characteristics (Kattel and Alldredge 1991, Kattel 1992). This research provides some key biological and harvest parameter estimates. In 2001, a research team from Simon Fraser University conducted a household survey in Phortse in order to collect information on social-capital attributes, and to identify the socio-economic feasibility of a CWM live capture and release project. This latter study was critical in developing the current model structure, and providing a number of parameters for both the CWM and PES strategies. Additional parameter estimates come from other musk deer studies in Nepal and other range states. Of particular note are musk deer studies from Kedernath National Park, India (Green 1985, Green 1987) and Baizha Forest in Qinghai province, China (Harris 1991, Harris and Guiquan 1993). Where gaps in parameter estimates exist for musk deer, other wildlife studies were used (Refer to the table of parameters in Appendix A for further details).

4 MODEL DESCRIPTION

This chapter provides further details of the model introduced in the previous chapter. Subsections include a general description (flow schematic), the community effort sub-model, the poaching and enforcement dynamics sub-model, the biological dynamics sub-model, and an overview of methods used to compare community wildlife management to payment for ecosystem services strategies.

4.1 General Description

Figure 3 shows the basic flow schematic for comparing PES and CWM strategies. Step 1 sets a number of economic, enforcement, stock, harvesting and poaching parameters as static for a given iteration of the model. Step 2 is a different process for the two conservation strategies. With respect to the CWM strategy, the next step (2a) is to solve for optimal enforcement and harvesting hours using a dynamic programming algorithm. The dynamic programming model returns 2 two-dimensional matrices that index the optimal harvest ($E^{\prime,*}$) and enforcement ($E^{\prime\prime,*}$) policies given a specific stock size. For the PES approach (step 2b), there are no harvesting profits so the annual subsidy directly determines the number of hours of enforcement. The enforcement hours, $E^{\prime\prime}$, for PES remains fixed each year given a constant annual subsidy z. After the optimal policies are saved, the last step is to simulate both models forward over a number of years, where the stock is sequentially subject to: (I) community harvesting dynamics, (II) community enforcing and outsider poaching dynamics (occur simultaneously), and (III) natural population growth of the musk deer population. The output of the forward simulation component includes equilibrium stock sizes and returns in profit. This framework allows for a comparison of CWM to PES strategies given a particular annual subsidy.

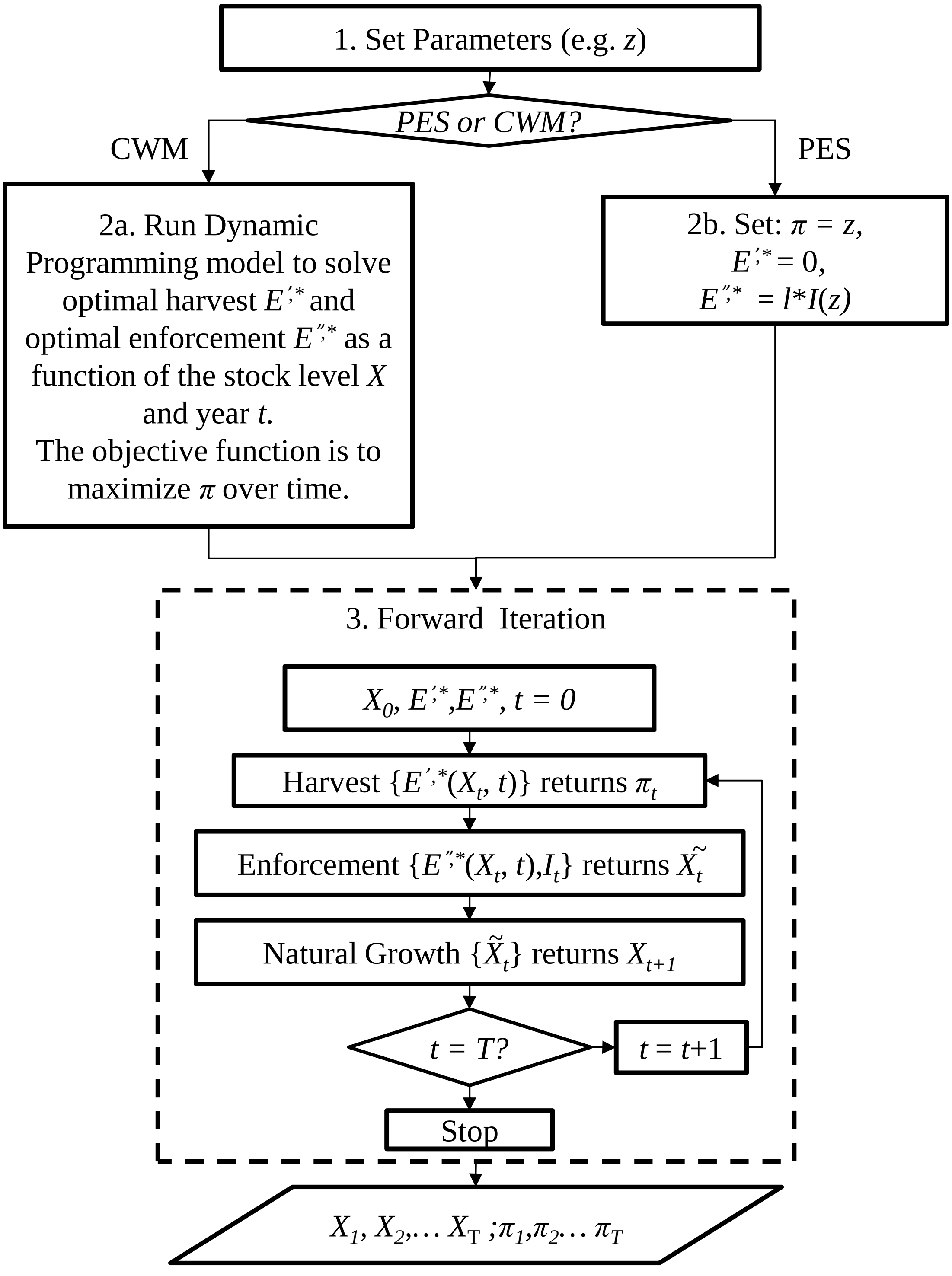

Figure 3. Flow model of analysis

4.2 Effort Sub-model

This sub-section provides details on equation 1.4. Recall that the number of participants who take part in either conservation strategy depends on the payment offer (i.e., profit per individual). To accommodate this specification, I make use of a 2001 household survey conducted in Phortse, Nepal (Wood and Knowler unpublished data). As part of the survey, participants were asked a contingent management question with respect to if a member from their household would be willing to provide 15 person days of labour towards a CWM capture-and-release project to collect musk and to patrol the forests against poachers in exchange for a hypothetical random payment. Several probit models were developed from the responses, basic household information, and social capital information (Wood and Knowler unpublished analysis). The probit models incorporated the following variables: payment offer, wealth status, level of participation in community groups/events, if the respondent was an executive member of a community group, participates in village decision-making, visits relatives outside the village, expressed level of trust of "others" outside the community, formal education, gender, if winter labour shortages are experienced, and the number of years living in the village. The probit model used in the simulation model incorporates statistically significant demographic variables with coefficients significant at the 5% level, and the payment offer significant at the 1% level. The probit model is:

$$\Gamma_t = \beta_0 + \beta_1 \pi_t + \beta_2 x_2 + \beta_3 x_3 + \varepsilon_t \qquad [2a]$$

where Γ_t is the predicted proportion of "Yes" responses, $\beta_0 = -4.489$, $\beta_1 = 0.0005$ is the coefficient for the payment offer π_t, $\beta_2 = 7.141$ is the coefficient for the variable *Wealth status* (x_2), $\beta_3 = -1.584$ is the coefficient for the variable *Group 3 member* (x_3), and ε is an error term with a standard normal distribution. The coefficients measure the change in z-score for a one-unit change in the associated variable. The variable *Wealth Status* is an aggregate of four household attributes representing a measure of a household's non-monetary wealth and the variable *Group 3 member* is a cluster of households sharing similar social-capital characteristics.[3] For the purposes of the simulation model, *Wealth status* and *Group 3 members* are set as the mean values reported in the household survey (Eq 2b). The cumulative normal distribution function, Φ, of the probit model for a given profit, π_t, provides the proportion of individuals willing to participate. Thus, the number of individuals willing to participate, *I*, is found by multiplying Φ with the total number of individuals available, *Y* (Equation 1.4).

$$\Gamma_t = (\beta_0 + \beta_2\bar{x}_2 + \beta_3\bar{x}_3) + \beta_1\pi_t = \beta_0' + \beta_1\pi_t \qquad [2b]$$

Returning to Equation [1.1], note that the terms π_t and I_t depend on one another. To resolve this problem of circularity, I assume individuals in the community have perfect information on the potential profit outcomes in the current year and are free to enter or exit the CWM project every year but the funds obtained in year *t* must be paid out to each individual participating that year (i.e., profit savings and deficits are not considered over multiple years). The possible number of male musk deer to capture is discrete, which results in a stepwise return of potential profits. The lines labelled π^T in Figure 4 represent iso-profits for a given number of male deer caught, *H*, which includes

[3] The non-monetary household attributes that make up *Wealth Status* include: number of agricultural fields owned by households, number of livestock, number of household members, and ownership of a tourism business. *Group 3 member* households are categorized as leaders, and share greater involvement in village decision-making and are more likely to have direct connections to people outside the community. Incidentally, individuals with higher *Wealth Status* are more likely to participate at lower payment offers, while *Group 3 members* are not.

the profit from the sale of musk and the annual subsidy. The total community profit is fixed along each π^T line and profit per individual, π_t, adjusts to the number of individuals participating. The vertical axis is shown as daily wage, π^{15}, and is an adjustment of profit per individual assuming that each participant commits 15 days of labour (i.e., $\pi^{15} = \pi_t/15$ days). The number of individuals that participate is determined where the line $I(\pi_t)$ crosses the line π^T (i.e., where the supply of I and demand for I is equal). More individuals would participate if π_t was higher, and less if π_t was lower. If the community does not capture any musk deer in year t, then the subsidy is the only source of profits.

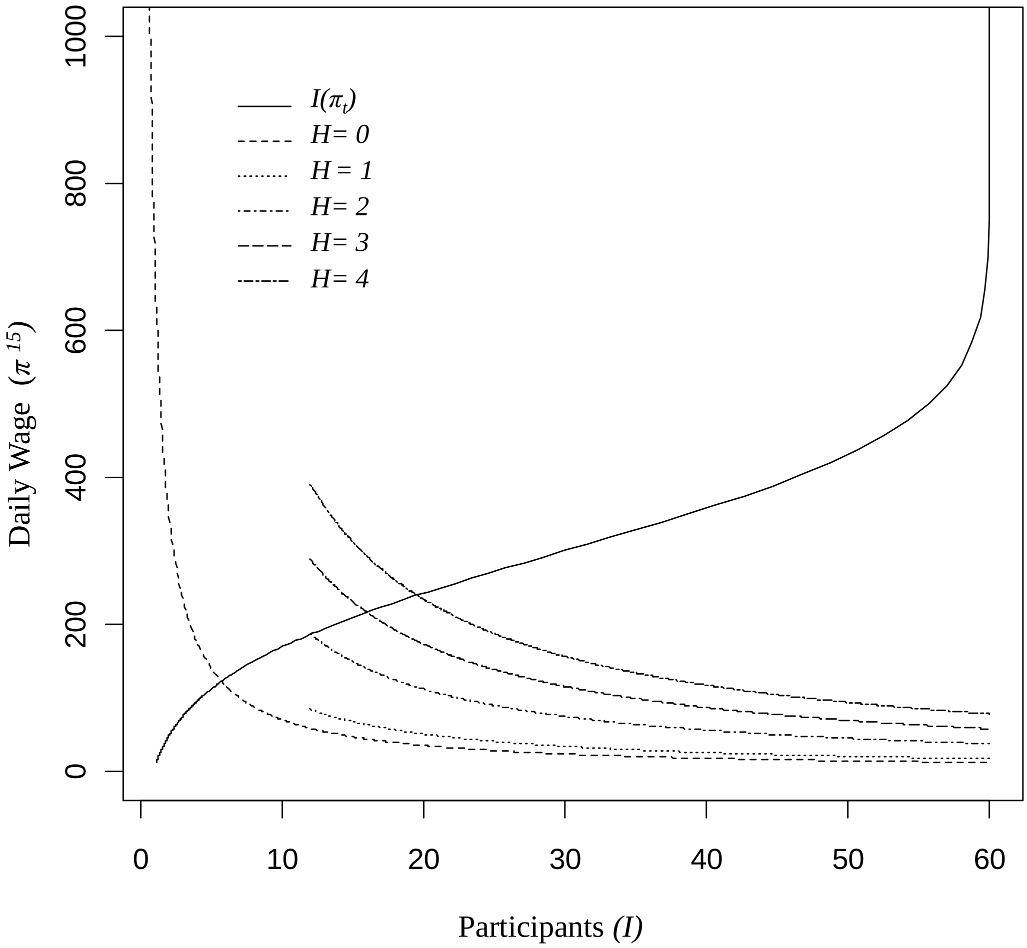

Figure 4. Labour supply curve, $I(\pi_t)$ (solid line), and iso-profit curves, π^{15}, given different number of male deer caught, *H*(broken lines). Units of daily wage, π^{15}, are in Nepalese rupees/day/participant.[4]

[4] Figure 4 depicts a fundamental economic concept of labour supply and demand. Labour supply is shown as the number of individuals, *I*, willing to participate given a payment offer, π^{15}. Derived labour demand is akin to the iso-profit curves, π^{15}, for a particular harvest level, *H*. The market equilibrium is where the two lines intersect.

4.3 CWM Harvest Sub-model

I assume the community selects the drive netting technique to harvest musk, which requires one to several participants to drive (i.e., frighten) the deer towards nets controlled by two to three participants each. Once a deer is captured, it is sedated for the musk extraction. After the deer recovers from the anaesthetic, the participants release it back into its environment. The process from capture to release takes approximately 45 minutes (Kattel 1992). Approximately 10 to 15 people are required to capture musk deer (Kattel 1992). Individual musk deer are assumed to be “harvestable” once a year.

The effort required to catch musk deer is based on a hyperbolic predator-prey functional response, whereby participants are limited in the number of musk deer they can capture due to handling time (Figure 5). Handling time includes the time to set up nets once a male musk deer is detected, the time spent driving the deer towards the net, the time it takes for the deer to recover from sedation, and the time spent unsuccessfully attempting to capture musk deer (Kattel and Alldredge 1991, Kattel 1992).

The length of time spent capturing deer, g_t, is:

$$g_t = \frac{E'_t}{Y^m} \qquad [3]$$

where, as indicated previously, E'_t is the total person hours allocated to harvesting in year t, and Y^m is the number of individuals required during a drive net capture.

Because musk deer with harvestable musk are gradually removed from the population *within* each harvest period the number of male musk deer caught should take on a discrete form as proposed by Hassell (1978: Appendix 1). Unfortunately, as Turchin (2003) points out, the number of male deer caught implicitly enters the formula on the RHS of Hassell’s equation making it difficult to solve. Instead, Turchin (2003) suggests the use of a continuous predator-prey model. In this case, assuming a hyperbolic response, the instantaneous number of musk deer caught per harvest group is given by:

$$\frac{dN(t)}{dt} = -\frac{\alpha_{\upsilon} N(t)}{1+\alpha_{\upsilon} h_{\upsilon} N(t)} \qquad [4]$$

where *N(t)* is the number of adult male musk deer with available musk at instant *t* (note the conventional use of *t* in brackets as opposed to its use as a subscript), h_{υ} is handling time, and a_{υ} is search rate. Equation 4 is solved numerically in the simulation model using a fourth-order Runge-Kutta algorithm over g_t hours with a time step Δt = 1 hour to determine the amount of musk deer harvested annually. Appendix B provides the derivation of parameter estimates for α_{υ} and h_{υ}.

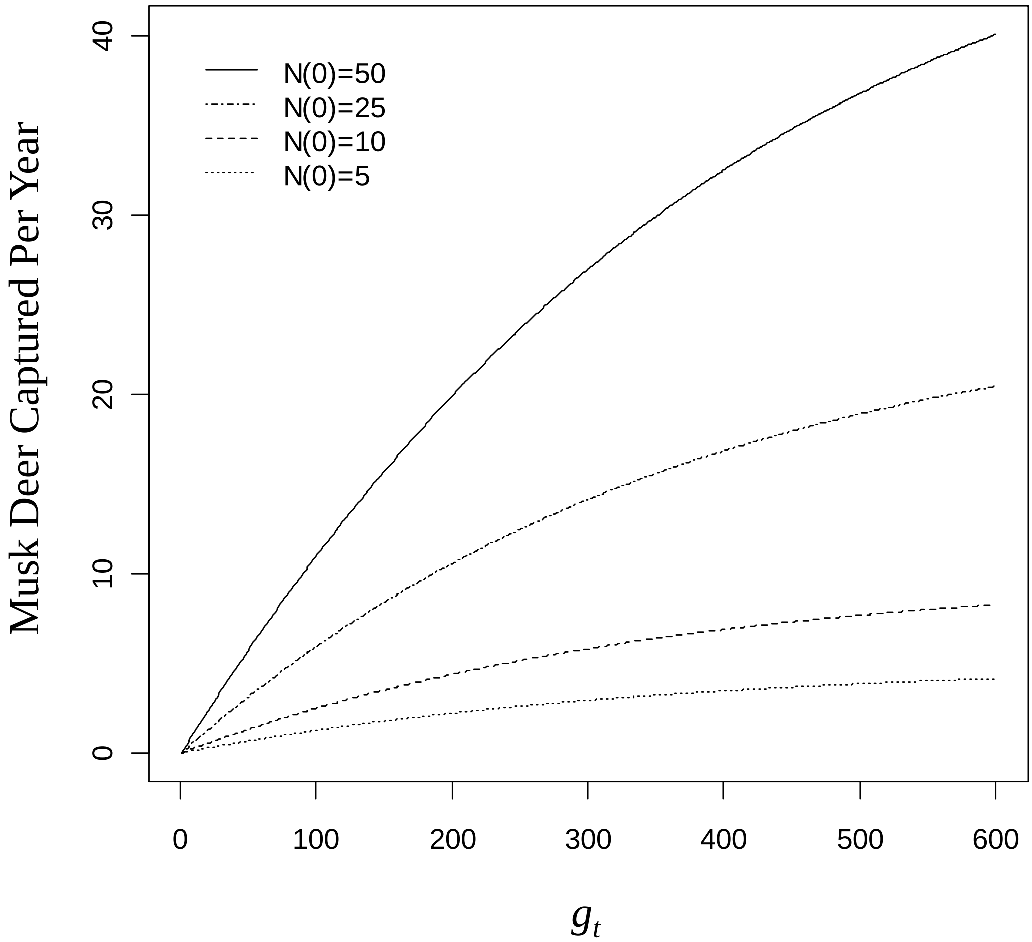

Figure 5. Musk deer caught based on number of capture hours available, g_t. The lines indicate different number of male deer available at the beginning of the harvest season (from 5 to 50/management area). The size of the management area is 10 km^2.

4.4 Poaching and Enforcement Sub-model

I assume poaching of musk deer comes from individuals/groups that originate from outside the community. Poachers have a high discount factor because they do not have recognized rights for the musk resource and therefore no incentive to manage the population over the long-term. From the poacher's perspective, if they do not exploit the population, then someone else will.

Various hunting methods that subsequently result in the death of the target deer would appeal to musk deer poachers due to their efficiency and reduced costs. A well-documented traditional hunting method includes the use of fire to drive musk deer towards poisonous spears (Jackson 1979). Other studies commonly report the use of guns (Harris and Guiquan 1993, Saberwal 1996). However, a ubiquitous and inexpensive approach is the use of snares (Upreti 1979, Green 1986, Rabinowitz and Khaing 1998, Mishra et al. 2006). Snares are small wires or nylon cords strategically placed in musk deer habitat to hold onto a deer's neck or leg. Deer captured by snares often die from strangulation. The use of snare traps is indiscriminate as it kills non-targeted animals including young musk deer, female musk deer and other non-targeted species – including other endangered species such as snow leopards and red panda (Green 1986, Rabinowitz and Khaing 1998, Theile 2003).

This analysis considers snares as the preferred technology for poachers.[5] If a community were to engage in anti-poaching effort, then poachers may put an emphasis on using snares as their preferred technology in order to avoid detection. In some forested areas of Ghana, an increase in enforcement effort led to a shift from the use of firearms to snares (Jachmann 2008). The authors of this latter study believe that snaring was preferred in order to avoid detection. Similarly, in Zambia, an increase in local enforcement efforts led to an increase in snaring as the preferred strategy in wildlife management areas (Gibson and Marks 1995).

[5] Future analysis could incorporate the use of firearms. However, since gun hunters can select the age and sex of the animal, and adult males are preferred, modelling would require additional information on the impact of reduced males to population growth.

Section 4.4.1 describes musk deer mortality due to the presence of snares, and how much snaring effort poachers decide to apply at the beginning of year *t*. Section 4.4.2 describes the interactions between poaching effort and enforcement effort within a particular year.

4.4.1 Poaching

The presence of snaring increases mortality in musk deer populations. Poaching deaths are modelled as:

$$P_t = X_t - \widetilde{X}_t(E_t'', I_t) \quad [5a]$$

where P_t is poaching deaths, X_t is the musk deer population in year *t* prior to poaching (note: this includes males, females and young because snaring is indiscriminate), and $\widetilde{X}_t$ is the number of individuals in year *t* that survive poaching, which depends on total person hours of enforcement, E_t'', and total number of individuals involved in CWM or PES in year *t*, I_t.

A typical scenario for snare trapping is that a group of poachers will enter an area and set snare traps for a few days to several weeks. After snares are set, poachers check their traps every day or every few days. For simplicity, I assume all snares are set instantaneously at the beginning of the poaching season. The assumption of instantaneous snare placement most likely puts a positive bias on the hazard from snares during the first few hours from a real situation because, in reality, snares would be set over time. However, the simplification enables a straightforward optimization of the allocation of anti-poaching effort (i.e., maximize I_t as discussed below). The change in musk deer numbers during a poaching season is modelled as:

$$dX(t)/dt = -\alpha s(t(E''_t, I_t))X(t) \qquad [5b]$$

where α is the mean encounter rate of an individual animal with a snare trap, and s is the number of snares active (i.e., a measure of poaching effort), which is a function of total person hours of enforcement, E''_t, and total number of individuals involved in CWM or PES in year t, I_t.

Very few studies specifically model poaching deaths due to snaring. Rowcliffe et al. (2003) develop a model to approximate off-take empirically reported for a number of species snared in Africa using a model similar to equation 5b. In their study, they assume that snares are placed randomly within an animal's home-range. Under this assumption, the mean encounter rate with snares, $\alpha_{(random)}$ equals $D\upsilon$, where: D is the total width of an individual animal's path it occupies perpendicular to its direction that could trigger a snare, and υ is the average velocity of the animal.

As opposed to random placement, poachers often set up fence barriers made from surrounding vegetation within musk deer habitat to direct musk deer through openings where snare traps are set (Oza 1988, Rabinowitz and Khaing 1998). Green (1978) estimated that snares set up in this fashion result in densities ranging from 100 to 600 snares/km^2 (Green 1986). Under this situation, the mean encounter rate used by Rowcliffe et al. (2003) no longer applies due to the non-random placement of snares. Therefore, I prepared a random walk simulation where deer are surrounded by these barriers to estimate a more suitable value for the mean encounter rate with snares, $\alpha_{(barrier)}$ (See Appendix B for a description of this model)

How much effort do poachers apply? Numerous analytical studies model poaching effort as a decreasing function of enforcement effort and an increasing function of wildlife stock assuming constant prices for off-take (Clarke et al. 1993, Damania et al. 2003). In these models, the poachers adjust their level of off-take in each period to optimize their welfare by avoiding penalties (e.g. fines and confiscation of poached items in their possession). I assume anti-poaching groups remove snares but do not actually capture poachers. Because poachers avoid detection, poaching effort is exogenous to the level of enforcement effort. For simplicity, snares are set at a constant density every year. A sensitivity analysis for the number of snares set will range between 100 and 600 snares/km^2. In the model, poachers adjust the length of time, *d*, they spend in the management area as a function of wildlife stock. I ignore a breakeven point (e.g., a switch to alternative income generating opportunity), because the sale of a musk from a single male deer can provide more than a quarter of a household's annual income (Jackson 1979, Khan et al. 2006). Assuming that poachers place no value on a future stock, because they lack rights to the resource, the intended poaching effort is to capture all adult male deer available. Thus, the target length of time poachers maintain snares, *d*, is determined by calculating the time it takes to deplete the adult male deer population, *N*, to 0.05 (essentially zero *N*). Thus,

$$d = \ln\left(\frac{0.05}{N(0)}\right) \Big/ \left(\frac{-\alpha s(t=0)}{24}\right) \qquad [6]$$

assuming poachers do not consider anti-poaching effort.

4.4.2 Enforcement

The poaching and enforcing activities occur simultaneously in the model. The change in the number of snares, *s*, is a function of the amount of enforcement effort over time. Search rates for discovering snares depend on the way snares are placed. If snares are placed randomly, then the search rate is likely longer in comparison to if they are placed along fence barriers. In the latter case, snares are easier to detect since the brush fences act as a visual cue for the presence of snares. In addition, once a brush fence is detected it is likely to contain multiple snares. As such, snares are removed by anti-poaching units with the following hyperbolic rate equation:

$$\frac{ds}{dt} = \frac{-\alpha_\tau \frac{s}{\bar{s}} I_t \bar{s}}{1+\alpha_\tau h_\tau \frac{s}{\bar{s}}} = \frac{-\alpha_\tau s I_t}{1+\alpha_\tau h_\tau \frac{s}{\bar{s}}} \qquad [7]$$

where α_τ is the rate of discovering snares with $\alpha_\tau = \alpha_{\tau(random)}$ for random placement of snares, $\alpha_\tau = \alpha_{\tau(barrier)}$ for groups of snares placed along fence barriers, s is the total number of snares, $\bar{s}$ is the average number of snare traps on a fence barrier that are found and removed once detected (if snares are placed randomly then $\bar{s}$ is set to 1) , I_t is the number of individuals engaging in anti-poaching effort and h_τ is handling time per snare ($h_\tau = h_{\tau(random)}$) or group of snares ($h_\tau = h_{\tau(barrier)}$). Equations 5 and 7 are solved simultaneously in R using a fourth-order Runge-Kutta routine with a time step equal to 1 hour, with the number of hours for anti-poaching, $g_t^{''}$, set as $g_t^{''} = E_t^{''}/I_t$. Ultimately, anti-poaching effort is more effective with more individuals, I_t, that can respond to snaring at a given instant, but also depends on the duration of enforcement (i.e., $g_t^{''}$).

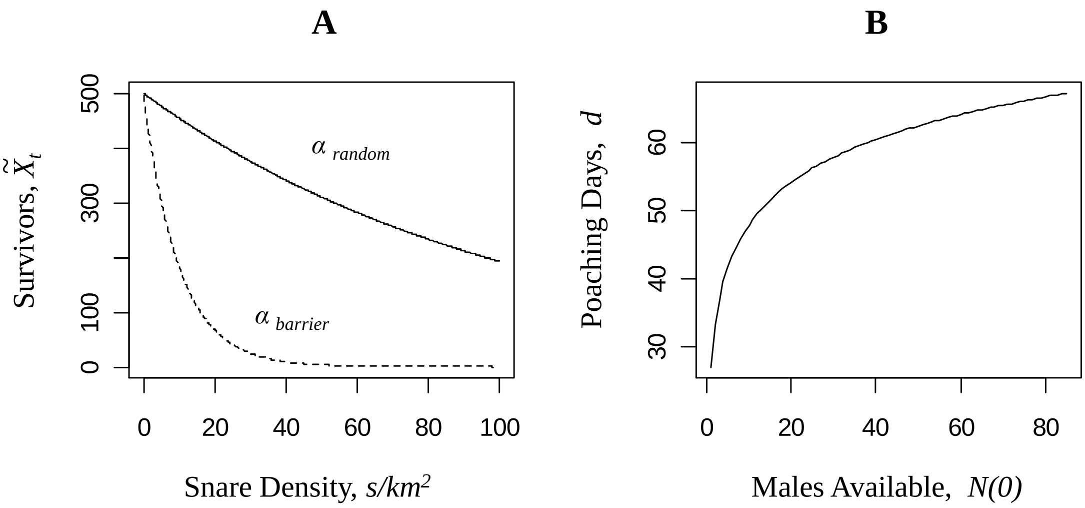

Figure 6. Poaching dynamics.

Panel A– Remaining population after exposure to 90 days of constant poaching pressure (i.e., constant snare density). Relationship based on parameter estimate from random walk model, $\alpha_{barrier}$ and from random placement of snares, α_{random} (i.e., Rowcliffe model). Panel B – Number of days that snares are active in order to reduce the male population to 0.05 male deer vs. number of males available at the beginning of the season.

4.5 Biological Sub-model

As shown in equation 1.2, population updates are a function of both natural growth and off-take from illegal poachers. Below are details for the natural growth component.

4.5.1 Natural Population Growth, *F*(*X*)

Natural population growth for musk deer is modelled by the theta logistic function:

$$X_{t+1} = \tilde{X}_t + r\tilde{X}_t \left[1 - \left(\frac{\tilde{X}_t}{K} \right)^{\theta} \right] \qquad [8]$$

where r is the intrinsic rate of growth, $\widetilde{X}_t = X_t - P_t$ is the stock of musk deer that remains after poaching activities occur, K is the carrying capacity, and θ is a shape parameter that determines when density dependent changes in growth occur relative to K. When $\theta > 1$, the per capita population growth rate is relatively constant and approximates r until $\widetilde{X}_t$ approaches K where it declines rapidly. If $\theta = 1$, then the per capita population growth rate declines linearly with greater $\widetilde{X}_t$. If $0 \leq \theta < 1$, then the per capita population growth rate declines more rapidly at low $\widetilde{X}_t$, and the decline in per capita population growth rate becomes more linear at higher $\widetilde{X}_t$ (Ross 2009). Given the effects of density dependent regulation are largely unknown for musk deer, the theta-logistic model tests for various density-dependent effects. High values of θ are suggested to occur with some ungulate populations due to limited resource availability at larger population densities (Saether 1997, Mayaka et al. 2004). However, other models of ungulate population dynamics apply the conventional logistic equation by setting θ to 1, which makes density dependence linear in population density. The conventional logistic model was able to closely approximate a data set of white tail deer in south-eastern Michigan (Jensen 1995). It was also applied to Saiga antelope populations in Russia and Mongolia (Milner-Gulland 1997).

In addition to uncertainty surrounding an appropriate value for the shape parameter, θ, no reliable estimate for r appears in the literature for any species of musk deer. A lack of a reliable value for r is due to a deficiency in time-series analysis for natural populations of musk deer. The value for r that I use as a baseline case is 0.194, and it comes from a partial life cycle model originally created by Oli and Zinner (2001). This partial life cycle model is based on skull data from a natural population of Forest

Musk Deer (*M. berezovskii*) (Yang et al. 1990). The r value is within the range of the 15 to 20% net population increase reported for a captive population of *M. berezovskii* (Parry-Jones et al. 2001). However, $r = 0.194$ is likely inaccurate due to a misapplication of skull data for estimating r (Caughley 1977, Harris and Metzgar 1993).[6] Furthermore, Himalayan Musk Deer (*M. chrysogaster*), likely have a lower intrinsic growth rate than Forest Musk Deer. The former species has a lower observed mean litter size in comparison to the latter species (Green 1989). I incorporate a sensitivity analysis for r due to the uncertainty of its value.

Since only adult males produce musk, their numbers are important to track in the model. The natural fraction of adults in the total population of musk deer that are male, f_m, is set to 0.17 based on reports from 105 carcasses of Himalayan musk deer that were indiscriminately culled with respect to age or sex in Nepal (Jackson 1979)[7]. However, the density of adult male musk deer is further limited by intra-specific competition whereby males establish territories within their home range that do not overlap with other males (Green 1985, Kattel 1992). A restriction in the number of males at high densities seemed a reasonable specification so as not to over-estimate potential earnings at higher population densities. As shown in equations 9.1 and 9.2, a constraint is applied during the population update so that if there are more adult males than available territories, then the adult male population, N_{t+1}, is limited to the total number of males the management area can accommodate, $\overline{N}$, and the total population, X_{t+1}, is adjusted as:

[6] In order for the observations of age-at-death from skull data to transfer into a schedule of survivorship parameters at different age classes (for use in a life table or a Leslie Matrix), *a-priori* knowledge of r is required in order to adjust observations over multiple cohorts to reflect a single stable cohort (Caughley 1977, Harris and Metzgar 1993). In their analysis, Yang et al. (1990) implicitly assume that r is zero. However, it is not possible to derive r from the skull data, the assumption of knowing r contradicts the possibility of solving for r.

[7] The percentage of adult males in the population is relatively close to 19% recorded for a captive population of Siberian musk deer (*M. moschiferus*) (Xu and Xu 2003).

If $N_{t+1} \geq \overline{N},$

then $$N_{t+1} = \overline{N} \qquad [9.1]$$

& $$X_{t+1} = \widetilde{X}_t + r\widetilde{X}_t\left[1\text{-}\left(\frac{\widetilde{X}_t}{K}\right)^{\theta}\right] - \left[\left[\widetilde{N}_t + r\widetilde{N}_t\left[1\text{-}\left(\frac{\widetilde{X}_t}{K}\right)^{\theta}\right]\right]\text{-}\overline{N}\right] \qquad [9.2]$$

The first two terms on the right hand side of equation 9.2 are the same as equation 8 and the third term removes the male population greater than $\overline{N}$ from the total population.

For the purposes of this study, the population is considered self-contained, no immigration or emigration is considered. The size of management area, A, is arbitrarily set to 10 km^2. The size is larger than the 1 km^2 forest originally estimated for the village of Phortse. However, a larger size was selected to compare for musk deer at lower carrying capacities, and because donors may not be motivated to intervene at too small of a scale. The carrying capacity, K, of the area is determined by K_b, the maximum density of musk deer (X/km^2). Based an extrapolation of pellet-count data (Musk deer faeces observations), un-poached populations of Himalayan musk deer have reached estimated densities of up to 71 individuals/km^2 in some locations (Liu and Sheng 2002). However, densities of 3 to 6 individuals/km^2 are more commonly reported for un-poached populations (Green 1985, Green 1986). Due to the wide range of musk deer densities found in the literature, an adjustment to the size of the K is part of a sensitivity analysis. The size of the stock will influence the success of the CWM strategy.

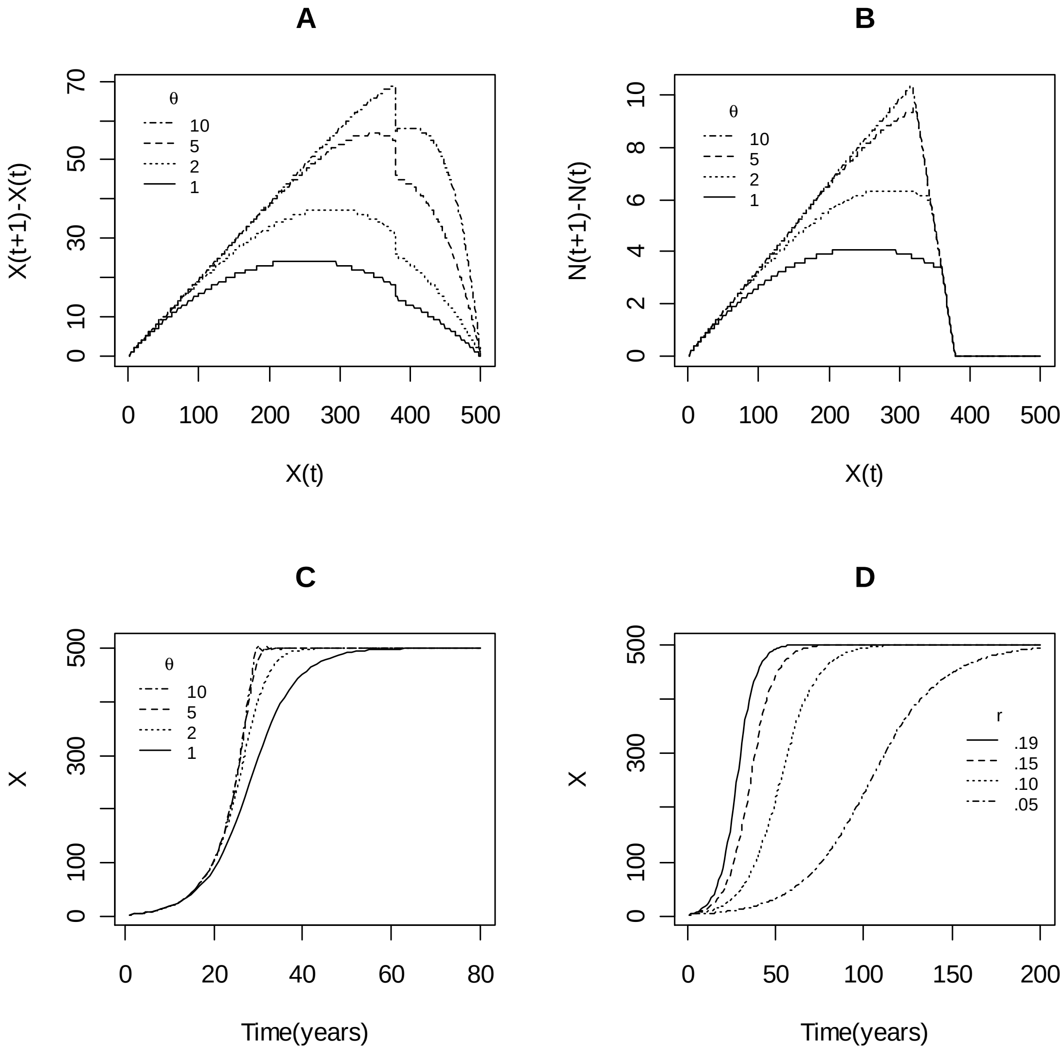

Figure 7. Effects of the shape parameter, θ, and intrinsic growth rate, r, on musk deer population. Panel A: Change in stock given current stock levels for various shape parameters. Panel B: Change in adult males given current stock level for various shape parameters. Panel C: Stock growth over time for various shape parameters. Panels A, B and C use baseline intrinsic growth rate (r = 0.194). Panel D: Growth over time with various r values and using baseline shape parameter (θ = 1). The 'kink' in Panels A and B reflects male territoriality limiting growth of males.

4.6 Comparison of Payment for Ecosystem Services and Community Wildlife Management Approaches

This subsection describes the approach taken to compare the two conservation strategies.

4.6.1 Payment for Ecosystem Services

As indicated in Section 3.1.4, I select subsidies to achieve stock equilibria of 50, 70 and 90% of K under the PES scenario. The subsidy required to achieve a particular equilibrium can be found by varying z and iterating the model calculations (Figure 3 - Step 3) until the appropriate I_t and $E_{\mathrm{t}}^{''}$ is found (Section 4.2 and Equations 1.4 to 1.6) that results in the target stock at time $t = T$. For the PES scenario, the forward iteration applies equations 5, 6, 7, and 9 to find X and π from $t = 0$ to $t = T$. The subsidies are then passed through the CWM model as described below.

4.6.2 Community Wildlife Management

The same set of subsidies, z, originally used in the PES model to achieve 50, 70, 90% of K are then passed through the flow schematic (Figure 3) under the CWM scenario for comparing economic and conservation outcomes. The enforcement and harvesting regime is now subject to community optimization (Equation 1.1). After passing through the dynamic programming model (Equation 2), optimal $E_{\mathrm{t}}^{'}$ and $E_{\mathrm{t}}^{''}$ policies are saved for each year and for each stock size. The CWM model is then forward iterated applying $E_t^{',*}$ and $E_t^{'',*}$. In addition to the equations used for PES, the CWM strategy also makes use of equation 4, which calculates the instantaneous number of musk deer caught per harvest group, to determine harvest levels.

4.6.3 Development and Conservation Performance Indicators

Development outcomes are compared by the net present value accruing to the community, while conservation outcomes are compared with resulting equilibrium stock sizes given the same subsidy for either strategy.

In the CWM scenario, the enforcement level depends on the different levels of stock (i.e., enforcement effort is not solely dependent on the subsidy as in the PES scheme). The surviving stock, $\tilde{X}$, is an indicator of the level of enforcement obtained through a PES or CWM subsidy. The value η compares the relative deer population response to a subsidy as:

$$\eta = \frac{\tilde{X}^{\mathrm{CWM},1} - \tilde{X}^{\mathrm{CWM},0}}{\tilde{X}^{\mathrm{PES},1} - \tilde{X}^{\mathrm{PES},0}} \qquad [10]$$

where $\tilde{X}^{\mathrm{CWM},1}, \tilde{X}^{\mathrm{PES},1}$ are the surviving stock levels in the *presence* of a subsidy at a given X for CWM and PES, respectively and $\tilde{X}^{\mathrm{CWM},0}, \tilde{X}^{\mathrm{PES},0}$ are surviving stock levels in the *absence* of a subsidy for CWM and PES, respectively. If $\eta = 1$, then the survival of stock from the presence of a subsidy is equivalent using either strategy, whereas $\eta < 1$ ($\eta > 1$) implies the survival of stock from the presence of a subsidy applying the CWM scheme is less (more) than the PES scheme.

4.7 Parameters

Base case parameters mostly come from a number of sources within the primary and secondary literature, while others were derived from secondary models (Appendix B). Refer to Appendix A for a table of parameters and sources (See also Section 3.2).

5 RESULTS

The purpose of this research was to compare a PES scheme to a CWM scheme for conserving musk deer. The first section of this chapter provides some general results for PES. The second section covers results for CWM and uses outcomes of PES as a point of comparison.

5.1 Payment for Ecosystem Services

The conservation outcome (i.e., equilibrium stock) of the Payment for Ecosystem Services scheme depends directly on the payment level provided by the donor. As a general rule, increases in payments lead to increases in equilibrium stock. As one would expect, labour constraints, biological parameters, and poaching dynamics can affect the expected costs of direct payments.

Since payments in a PES scenario directly fund enforcement, the enforcement effort is constant year-to-year assuming that the donor provides a constant annual subsidy, which results in a proportional survival rate with respect to population of musk deer (Figure 8, line *P*). If the subsidy increases (decreases), then the line *P* rotates right (left) through a fixed point at 0, 0 (rotation not shown). For a subsidy set at a target of 50% of *K* and with a shape parameter $\theta = 1$ a global equilibrium results at X ≈ 250 (Figure 8, equilibrium a). If the shape parameter $\theta = 10$, then the same subsidy would result in a greater equilibrium at X ≈ 480 (Figure 8, equilibrium b). However, if $\theta > 1$, then multiple equilibria could exist because portions of the natural growth rate are also linear for a given range of stock (e.g., consider if line *P* rotates left (not shown) and lies along the straight portion of the line $f(\widetilde{X})$, θ =10 between X = 0 and X ≈ 300). Under the PES scenario, the survival of stock increases with increasing levels of payment up to the point where the available hours among participants, E_t'', reaches the labour constraint, $\overline{E}$ or to the point where the stock $X = K$ (not shown). Further payments beyond this level are ineffective, as increased payments do not attract any additional participants or cannot increase the stock.

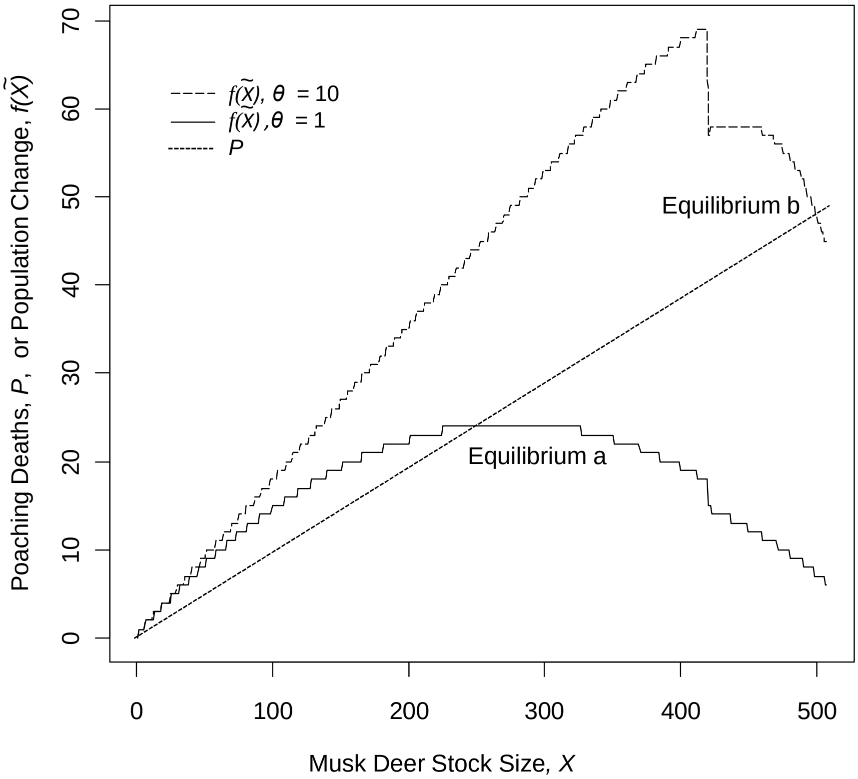

Figure 8. Poaching deaths, P (dotted line), and population change, $f(\widetilde{X})$ for θ = 1(solid) and θ = 10 (dashed), as a function of musk deer stock size, X, at start of year under a payment for ecosystem services scheme. In this example, the subsidy was set to target a stock equilibrium of 50% of K when shape parameter θ = 1 (i.e., poaching deaths and population change are equal at equilibrium a). However, if the same subsidy is applied when the shape parameter is higher, θ = 10 then the stock equilibrium settles at a higher level (equilibrium b). The abrupt drop in population change, $f(\widetilde{X})$, near a stock size of 400 occurs because of limits to adult males. Once territories are fully occupied by adult males, additional males are suppressed due to model constraints specified in equation 9.1 and 9.2.

Figure 9 shows equilibrium levels at $X_{t=80}$ for various daily wage rates under the PES scenario. The payment required to achieve a desired equilibrium depends on the input parameters. From numerical simulations with a target of (say) 70% of K, the annual subsidy and the number of participants required can range widely depending on parameters (Table 1). Favourable biological parameters, high r or high θ, lead to greater stock recovery after poaching deaths, and this lowers the cost of enforcement to achieve a desired equilibrium (Table 1, Figure 9). Interestingly, under particular circumstances, lower rather than higher payments may result in higher equilibrium stock levels after the natural population update, because the population overshoots carrying capacity at higher levels when the surviving stock, $\widetilde{X}_t$, is sufficiently below K (line $\theta = 10$ in Figure 9) .

A large initial snare density increases the cost of enforcement substantially. If poaching pressure increases (such as to $s(0) = 600$ snares/km^2 - an increase from the base case by a factor of 6), then the costs of enforcement increases by an even larger amount (i.e., a factor of 20 to achieve a stock size of 250, and a factor of 30 to achieve a stock size of 350) (Table 2). This is partially due to the assumptions made on the poaching sub-model because many deaths occur within the first few hours of the poaching season, and thus a larger level of anti-poaching participation is required upfront.[8]

[8] Even if snares entered the space more gradually, more anti-poaching effort would inevitably be required with increases in snares. However, further analysis specifying a gradual snare placement dynamic is required to explore how it would affect anti-poaching requirements.

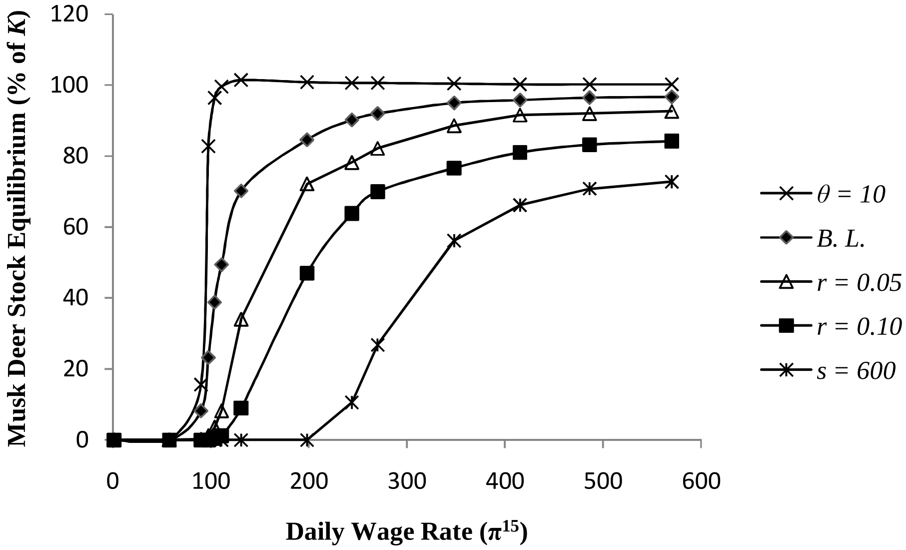

Figure 9. Sensitivity analysis of musk deer stock equilibria at t = 80 for different daily wage rates under the PES scenario. *B. L.* = baseline parameters (i.e., r =0.194, s = 100/km^2, and θ = 1, K = 500).

Table 1. Sensitivity analysis of selected parameters: intrinsic growth (r), shape parameter (θ), snare density (s/km^2), and number of individuals available in the community to participate in PES (Y). Responses to parameters shown as change in subsidy(z), number of participants (I), and daily wage (π^{15}) required in order to achieve a musk deer stock equilibrium of 70% of K under the PES scenario.

Parameters				*Subsidy, z, required to reach 70% of K*	*Individuals participating, I*	*Daily wage rate, π^{15}*
Intrinsic growth, r	*Shape parameter, θ*	*Snare density, s/km^2*	*Number of individuals available to participate in PES, Y*			
.194	1	100	60	12182	6.3	130
<u>0.10</u>	1	100	60	34452	12.2	188
<u>0.05</u>	1	100	60	104922	25.5	274
.194	<u>10</u>	100	60	5394	3.87	93
.194	1	<u>600</u>	60	375000	53.5	467
.194	1	100	<u>80</u>	10460	6.4	109

5.2 Community-based Wildlife Management Outcomes in Comparison to Payment for Ecosystem Services

In contrast to PES, the CWM scenario has a second line of production (harvesting). Therefore, the conservation outcomes in response to a subsidy are not as straightforward.

5.2.1 Conservation and economic outcomes under favourable resource conditions with high stock capacity, K_b = 50, K = 500.

Figure 10 displays time series projections of the total population of musk deer with respect to base case parameters (also refer to Table 2). If there is no subsidy, then the stock quickly drops to zero for the PES strategy (not shown). However, a zero subsidy has two possible results for the CWM scenario. The stock approaches X =379 (~76% of K) if $X_0 \geq 95$ and declines to zero if $X_0 < 95$. The presence of a non-zero equilibrium stock size, without a subsidy, suggests that a CWM strategy is financially sustainable under baseline conditions provided the initial stock size is sufficiently high (i.e., the CWM project can self-finance its operational costs between 95 and 379). As designed, the stock reaches target levels of 50, 70 and 90% of carrying capacity with increases in the annual subsidy under the PES strategy. In contrast, in the CWM strategy the stock size approaches 76% of K regardless of the three levels of subsidy. An increase in subsidy, does however, slightly reduce the time for the stock to reach the equilibrium in the CWM strategy.

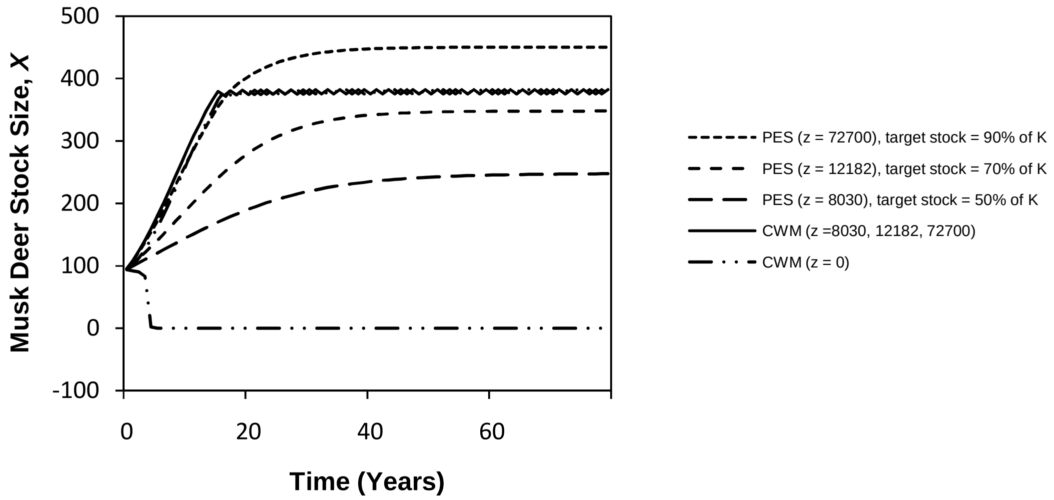

Figure 10. Trajectory of stock over time for PES and CWM strategies at different annual subsidies. Notes: (1) Solid line is combined trajectories for the annual subsidies, z = 8030, 12182, 72700. (2) The stock is initiated at X_0 = 94 and X_0 = 95 for z = 0. Depending on initial stock conditions, X_0, the CWM strategy with no subsidy has diverging trajectories (i.e., X increases if $X_0 \geq 95$, and decreases if $X_0 < 95$).

Table 2. Conservation and development outcomes with baseline parameters at high stock level (K = 500, K_b = 50 X/km^2). Columns 2 and 3 show the conservation outcomes for PES and CWM respectively as stock equilibria under different subsidies (Column 1). Column 4 shows the percent range in stock where CWM does not require a subsidy to achieve a viable population. Column 5 and 6 show development outcomes as Net Present Value (NPV) in Nepalese Rupees (NPR) accrued to the community. Column 7 shows the ratio of development benefits accrued by CWM in comparison to PES.

	Annual subsidy (z)	Stock Equilibrium: PES	Stock Equilibrium: CWM	CWM, Self sufficient range with no subsidy.	Community Total Profit (NPV): PES/ Donor Cost	Community Total Profit (NPV): CWM	Ratio of NPV = CWM/PES
Column #	1	2	3	4	5	6	7
Units	*NPR*	*% of K*	*% of K*	*% of K (min to max)*	*NPR (1000s)*	*NPR (1000s)*	*Col #6/Col #5*
	0	0	76 *(75-76)*	19-76	0	3,485	-
	8030	50			75	3,720	49.63
	12182	70			114	3,787	33.31
	72700	90			678	4,609	6.79

Further exploration of harvesting and enforcement dynamics help to explain the conservation results. First, consider when *no* subsidy is present. If the stock is below 14.3% of *K*, then the community has no interest to engage in either harvest or enforcement effort (Figure 11B and 11D). At stock levels between 14.3% and 18.8% of *K*, the stock is high enough to encourage harvesting (Figure 11B), but is too low to encourage sufficient enforcement effort and net population change remains negative (Figure 11A). Thus, the community rapidly liquidates the population between these stock levels. At stock levels greater than 18.8% of *K*, the population is high enough to generate profits attracting participants to engage in both harvesting and sufficient enforcement for sustained populations over the long term (Figure 11A and 11B).

If a subsidy is present for CWM strategy, then the community optimally allocates the majority of funds towards enforcement at lower stock sizes, <19% of *K*, so that the poaching off-take closely approximates the PES strategy (Figure 11A and 11C). Incidentally, the community also begins harvesting at lower stock levels with the presence of a subsidy (Figure 11B). However, the poaching off-take quickly approximates the zero-subsidy case once the stock is within the economically viable range (Figure 11A and 11C). Donor payments would be redundant between a stock size of 19% and 76% of *K*. However, should an international donor desire stock levels greater than 76% of *K*, they would need to implement a PES strategy because enforcement is not responsive to additional funds in the CWM strategy beyond this stock level.

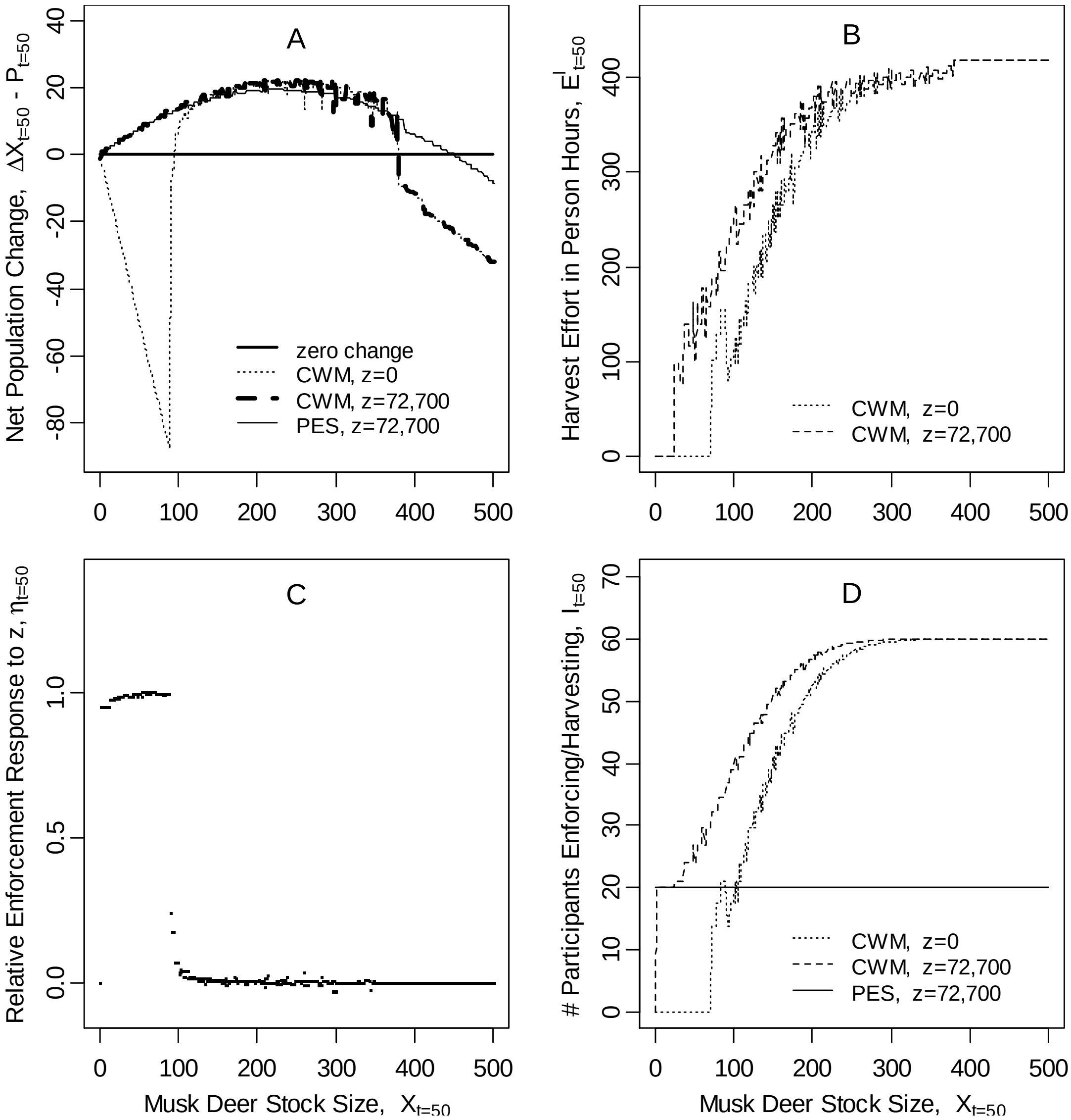

Figure 11. Details of optimization policy in year 50 over range of stock levels with base case parameters, with subsidies set at *z* = 0 NPR and *z* = 72,700 NPR (i.e., latter subsidy set to achieve a target musk deer stock equilibrium = 90% of *K* for PES strategy). Panel A: Optimal net population change, as a measure of the enforcement policy for CWM and PES strategies. Panel B: Optimal harvest hours, *E′* for CWM strategy with and without subsidy. Panel C: Relative community anti-poaching effort response to a change in subsidy (i.e. from zero to 72,700 NPR) between CWM and PES Strategies. *η* calculated from Equation 10. Panel D: Number of individuals, *I*, participating in a CWM or PES strategy.

Given the same annual subsidy, the number of participants, I, in a CWM scheme is equal to or higher than PES scheme (e.g., see Figure 11D). Results with the use of base case parameters represent a special case where the labour constraint sets in near a stock size of 60% of K for CWM (Figure 11D).

Higher participation, I, is more successful in responding to snares. As such, a CWM strategy may have a comparative advantage to PES at removing snares because participation under a CWM scenario secures equal-to or greater participation than PES due to additional streams of revenue. However, greater enforcement is not necessarily the case. For example, as shown in the baseline case, a higher I due to an increase in subsidy allows for a reduction in E_t'' without compromising survivors $\widetilde{X}_t$, as a trade off occurs between I and E'' (i.e., a higher I and a lower E'') for CWM. The resulting poaching deaths, P, or survivors, $\widetilde{X}_t$, with *or* without the subsidy was similar for stocks greater than 19% of K (Figure 11A and 11C). Instead, participants maximize their utility by allocating extra effort towards E_t' (Figure 11 B).

Table 2 provides results for the base case scenario. As expected, the economic returns received from CWM are always equal or larger than returns received from PES since there is an additional source of revenue. The CWM strategy appears to generate relatively high economic development opportunities under the base case conditions. For instance, if the initial stock size (X_0) is set at 20% of K (i.e., 100 individuals), then the net present value (NPV) of total profits entering the community ranges between 3.5 and 4.6 million Nepalese Rupees (i.e., USD 49K to 65K) depending on the annual subsidy provided. Assuming the same annual subsidy and given base case parameters, economic returns from the CWM strategy are 7 to 50 times greater than revenues received from a PES strategy.

Table 3 provides a sensitivity analysis for a variety of parameters when the carrying capacity of the management area is high (K = 500, K_b=50). Changes in biological parameters influence the relative effectiveness of the two strategies (Table 3). The range of stock levels that a CWM strategy can maintain without an annual subsidy diminishes slightly when the intrinsic growth rate, r, decreases (i.e., from 19% to 76% of K for the base case to 21% to 76% of K for r = 0.05). In comparison, a high shape parameter, θ, which maintains higher growth rates at higher densities, can result in a rather large jump in the equilibrium stock size for CWM (i.e., up to 100% of K for θ=10). If the constraint on male territoriality, $\overline{N}$, is removed, then the equilibrium with no subsidy expands to a new upper limit with a mean of 96% of K under the CWM strategy (See $\overline{N} = f_m K$ in Table 3). On the other hand, if $\overline{N}$ is lower (i.e., n = .35, $\overline{N}$ = 28), the equilibrium drops to 36% of K. This reveals that the optimal stock size is not at max ΔX, as in many other renewable resource models, but closely approximates the maximum number of adult males, $\overline{N}$.

Based on model assumptions, a larger pool of potential labourers, Y, would result in lower costs to attract the same number of participants, I, because more households are willing to participate at lower payoffs (assuming the spread of the probit model extrapolates to larger populations). As such, both PES and CWM strategies benefit from a larger Y. However, an increase in the stable equilibrium of the CWM strategy also leads to a greater factor increase in total profits for the CWM strategy (Table 3).

An adjustment of the discount rate, δ, has minimal effect on conservation outcomes (Table 3). The trade-off between harvesting in the current period or in the future is less severe for live-capture-and-release models in comparison to consumptive wildlife resource models. More harvests in the current period does not limit harvesting in the future. Instead, more harvesting can lead to greater population growth as the community has a larger profit and the financial capacity to ramp up its enforcement effort.

A key assumption required for CWM to appear financially lucrative at high stock densities, K_b, is that the production function for CWM also benefits from a rather high success rate of capture - as parameterized from a study by Kattel (1992).[9] However, some musk deer populations are more cautious of humans and are likely to flush at greater distances from a perceived threat, making capture effort much more difficult (Harris and Guiquan 1993). If the effective search rate, α_υ, takes on a lower value, and handling time, h_υ, takes on a higher value due to a low success rate of capture, at levels based on a study from Harris (1993), then the benefits of CWM disappear nearly completely (Table 3).

If the harvest team's speed changes, then the effective success rate of capture also changes proportionally. For example, if v_υ drops by half, from 1.34km/hr to .67 km/hr, then the effective search rate, α_υ, also drops by half (i.e. from .003 to .0015). A decline in the effective success rate of capture can substantially change the benefits of CWM. If speed of the harvest group drops by half then the minimum density of musk deer required to operate without a subsidy rises from 19% to 35% of K. At the same time, the relative

[9] Refer to Appendix B for a description on how "success rate of capture" enters into the harvest dynamics.

benefit of CWM to PES drops from 49 to 9 fold because of the decline in the number of deer harvested.

The speed of the musk deer, v, affects the instantaneous death rate of deer per snare trap, α, and thus the cost of community enforcement. I conducted a sensitivity analysis on instantaneous death rate due to uncertainty on the value of mean musk deer speed in the base case scenario. The base case value for mean deer speed could be low because it was derived from an observation of musk deer movement in snowy conditions which could have inhibited deer movement (Green 1985). If the average speed of deer is twice as fast as the base case scenario, then the cost of enforcement is approximately 4 times higher in order to achieve a stock equilibrium of 50% of K (Table 3, α = 1.41 e^{-05}). If the average speed is three times as fast, then the cost of enforcement is nearly 10 times higher (Table 3, α = 2.55 e^{-05}). An increase in the cost of enforcement reduces the relative benefits accrued to CWM in comparison to PES because more effort is allocated to enforcement.

The price of musk is set to the current market price (i.e. 45,000 USD/kg) in the sensitivity analysis, assuming that the community would receive the full benefit of the sale of musk and eliminate any “middle-man”. The base case price was set to reflect prices assumed in previous literature (Wood et al. 2008). In the model, this upwards adjustment equates to a price of p = 41,414 Nepalese Rupees or 583 USD per musk deer captured per year. The CWM scenario is lucrative when there are high economic returns, even at low densities of musk deer. The community would be willing to operate a CWM without a subsidy at 8 % of K (Table 3). The relative economic benefit of CWM to PES is 122 fold when the target stock is set to 50% of K (Table 3).

Table 3. Conservation and development outcomes of sensitivity analysis at high stock level (K = 500, K_b = 50 X/km^2)

Parameter*	Annual subsidy (z)	Stock Equilibrium: PES	Stock Equilibrium: CWM	CWM, Self sufficient range with no subsidy.	Community Total Profit (NPV): PES/ Donor Cost	Community Total Profit (NPV): CWM	Ratio of NPV = CWM/PES
Column # 1	2	3	4	5	6	7	8
Units	*NPR***	*% of K*	*% of K****	*% of K (min to max)*	*NPR (1000s)*	*NPR (1000s)*	*Col #7/Col #6*
δ = 0	8030	50	78 *(75-81)*	19-78	642	55,370	86.19
δ = 6	8030	50	76 *(75-76)*	19-76	141	9,201	65.48
δ = 16	8030	50			58	2,483	42.66
	47746	51			446	2,179	4.89
r = .05	104521	71	76	21-76	975	3,008	3.08
	238196	80			2,223	4,490	2.02
	17277	49			161	2,401	14.89
r = .10	34452	70	76	20-76	322	2,724	8.47
	199560	89			1,862	4,753	2.55
	0	0			0	1,497	-
θ = 10	5080	29	101	20-101	47	4,057	85.58
	5394	73			50	4,061	80.68
$\bar{N}$ = 28	0	0	36 (*35-38*)	20-38	0	2,087	-
$\bar{N}$ = 87	0	0	96(95-96)	20-94	0	3,630	-
s= 600	166000	50	72	25-72	1,549	4,424	2.86
	375000	70			3,500	6,572	1.88
	6620	49			62	4,198	67.96
Y = 80	10460	71	*79(75-83)*	18-82	98	4,254	43.58
	50300	88			469	4,831	10.29
	4583	55			43	5,053	118.15
Y = 120	7641	76	97	15-97	71	5,096	71.46
	40507	90			378	5,581	14.76
p = 41,141	8030	50	76	8-76	75	9,152	122.03
	8210	49	52		77	77	1.00
α_υ = .000022 h_υ = 10.75	12604	70	70	NA	118	118	1.00
	74000	90	*60(42-79)*		691	710	1.03
α_υ = .0015	8030	50	*77(75-78)*	35-76	75	703	9.37
α = 1.41 e^{-05}	31636	50	*77(75-78)*	19 -76	295	3,781	12.81
α = 2.55 e^{-05}	77892	50	*76(74-78)*	20 -76	727	4,085	5.62

*Stock initialized, X_0, at twenty percent of carrying capacity, K. **NPR = Nepalese Rupees. ***Mean (minimum to maximum) equilibrium. ****Baseline parameters: Maximum pool of participants/households Y = 60; snare density s = 100/km^2; capture rate α_υ = 0.003, handling time h_υ = 1.5 hours/N caught; shape parameter θ = 1; instantaneous death rate per snare α = 4.63e^{-06}; price of musk per musk deer received by community p = 19,350 intrinsic growth r = .19; discount rate δ = 12%; and adult male capacity $\bar{N}$= 64.

5.2.2 Conservation and economic outcomes when resource conditions are relatively poor (i.e., lower stock capacities).

Many natural populations of musk deer exist at lower densities than those reported for Phortse, Nepal. Thus, this section considers the conservation and development outcomes of both CWM and PES schemes with lower musk deer densities. When lower densities of musk deer prevail, then pursuing a CWM strategy is not as worthwhile as was observed at higher densities. Figure 12, compares the stock trajectories for both CWM and PES strategies of three subsidies when musk deer density k_b =10 km^2 and all other parameters are at baseline levels (see also Table 4). Once again, subsidies are set to obtain 50, 70 and 90% of K for PES. However, CWM can no longer maintain a population over time without a subsidy, even when initiated from a stock at K (notice negative net population in Figure 13A for CWM, z = 0). If stocks commence at 20% of K and the subsidy is set achieve a equilibrium stock of 50% of K for PES, then the stock trajectories for PES and CWM are identical. The optimal split of labour for CWM is to put all effort into enforcement. At a subsidy that targets 70% of K for PES, the stock trajectory exhibits a pulsing behaviour in the CWM scenario (Figure 12). Periodic harvests cause enforcement effort to drop during years of harvesting.[10] At a target of 90%, the stock trajectory of CWM remains slightly lower than PES, but the community benefits from additional profits due to annual harvests (Figure 12). The stock trajectory is lower because there is a preference for harvesting, especially as the stock approaches K (Figure 13B,C)

[10] Note: Rondeau and Conrad (2003) also observe pulses in harvesting as an optimal strategy - but in controlling white tail deer populations).

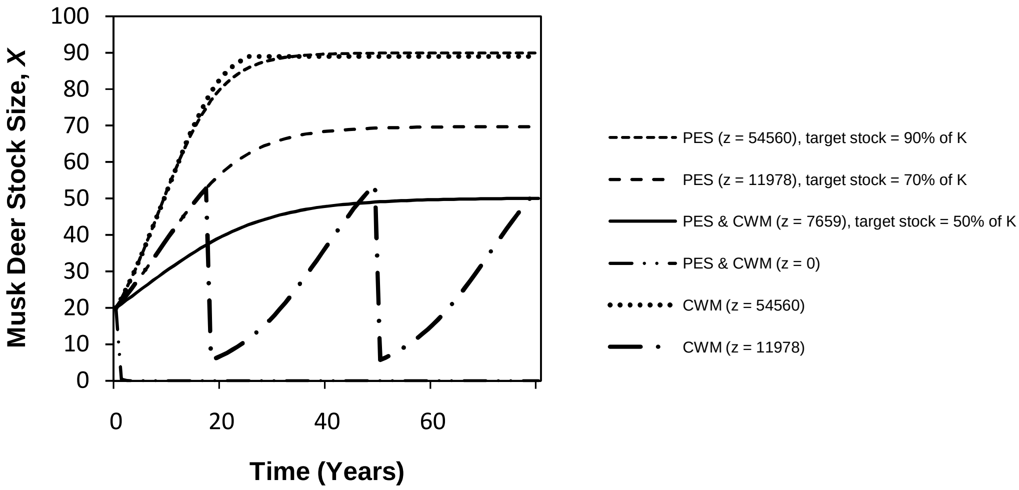

Figure 12. Trajectory of stock over time for PES and CWM strategies at different annual subsidies under low stock capacity (K_b = 10).

Table 4. Conservation and development outcomes with baseline parameters at low stock level (K = 100, K_b = 10 X/km^2)

	Annual subsidy (*z*)	Stock Equilibrium		CWM, Self sufficient range with no subsidy.	Community Total Profit (NPV)		Ratio of NPV = CWM/PES
		PES	CWM		PES/ Donor Cost	CWM	
Column #	1	2	3	4	5	6	7
Units	*NPR***	*% of K*	*% of K****	*% of K (min to max)*	*NPR (1000s)*	*NPR (1000s)*	*Col #6/Col #5*
	0	0	0	NA	0	0	-
	7659	50	50		71	71	1.00
	11978	70	26 (6-53)		116	118	1.02
	54560	90	89		547	779	1.42

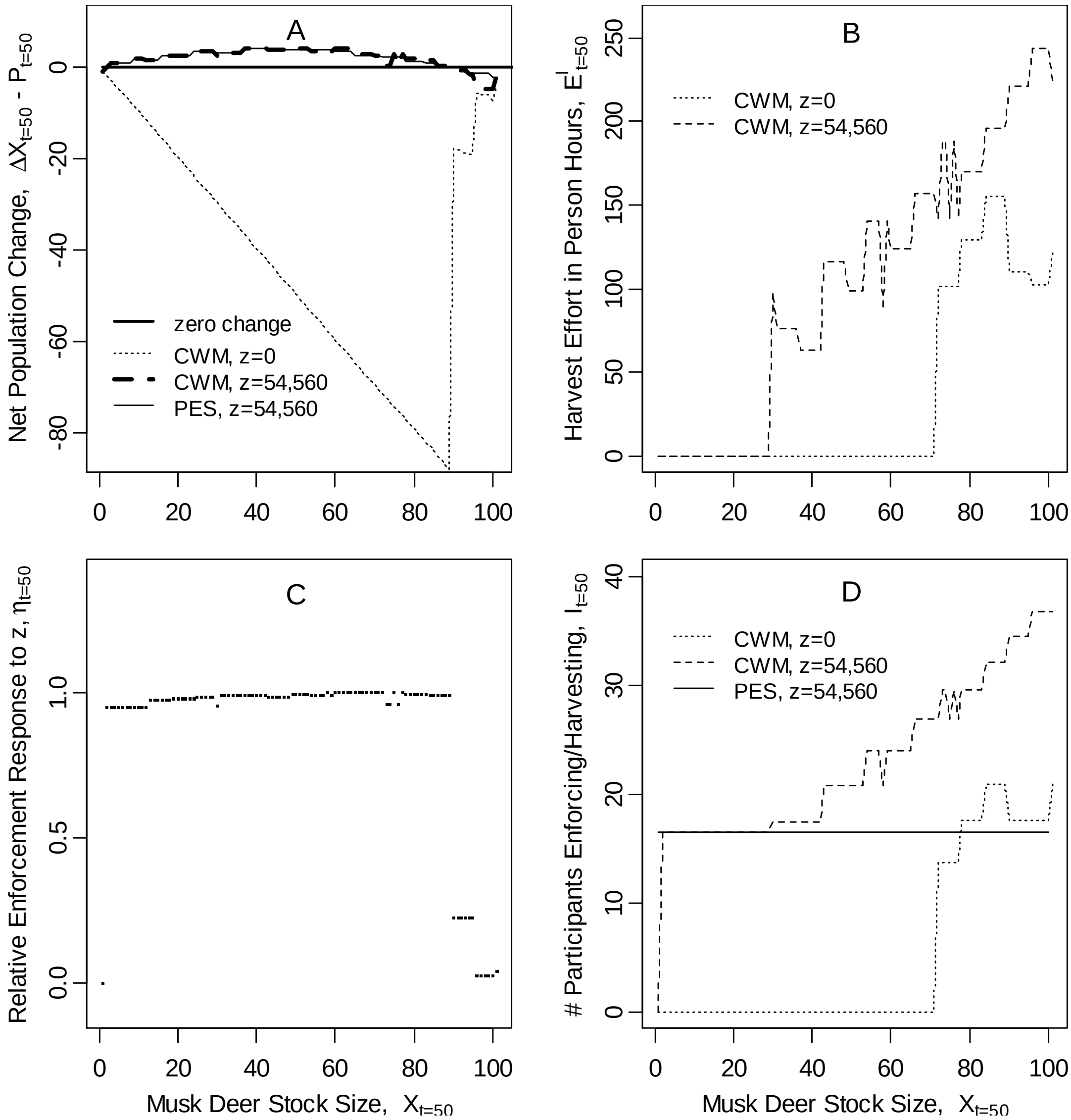

Figure 13. Details of optimization policy in year 50 for all possible stock levels with base case parameters, with the exception of a low musk deer density, K_b = 10, and subsidies selected at z = 0 and z = 54,560 NPR (i.e., target equilibrium stock of 90% of K for PES). Panel A: Optimal net population change, as a measure of the enforcement policy, for CWM and PES strategies. Panel B: Optimal harvest hours, g, for CWM strategy with and without subsidy. Panel C: Relative community anti-poaching effort response to a change in subsidy (i.e. from zero to 54,560 NPR) between CWM and PES Strategies. η calculated from Equation 10 Panel D: Number of individuals participating in CWM or PES.

Table 5 shows results from a sensitivity analysis for a selection of parameters at low stock levels with respect to conservation and development outcomes. Of note, is the low potential for CWM at $K_b = 5$ and 10. Recall that many un-poached musk deer populations in Nepal likely exist at low population densities between 3-6 X/km^2. When all other parameters are at baseline conditions, including favourable r and α_v values, the equilibrium of X for CWM is equal or less than the equilibrium for PES. Furthermore, there is no financially self-sufficient range for CWM (i.e., a subsidy is required to maintain an equilibrium of $X > 0$) (Table 4). However, if the minimum labour constraint for CWM is relaxed, say $Y^m = 4$, then the conservation and development outcomes appear to make CWM more attractive (Table 5). This suggests that an improvement in capture efficiency could make CWM more financially viable at lower stock sizes.

Table 5. Sensitivity analysis at lower stock capacities (K_b = 5, 10, 15 and 20 X/km^2)

Parameter*		Annual subsidy (z)	Stock Equilibrium PES	Stock Equilibrium CWM	CWM, Self sufficient range with no subsidy.	Community Total Profit (NPV) PES/ Donor Cost	Community Total Profit (NPV) CWM	Ratio of NPV = CWM/PES
Column # 1		2	3	4	5	6	7	8
units		*NPR***	*% of K*	*% of K****	*% of K (min to max)*	*NPR (1000s)*	*NPR (1000s)*	*Col#6/ Col#7*
$K_b = 5$		7480	50	50	NA	70	70	1.00
		12000	70	70		112	112	1.00
		55000	90	88		513	540	1.05
$K_b = 5$, $\theta = 10$		4740	60	60	NA	44	44	1.00
		4900	80	80		46	46	1.00
		5080	92	92		47	47	1.00
$K_b = 10$	r =.10	17044	50	50	NA	159	159	1.00
		34740	70	65 (64-65)		324	333	1.03
		201300	91	90		1879	2169	1.15
	r =.10, X_0=K	201300	91	90	NA	1879	3327	1.77
	θ=10	4900	50	50	NA	46	46	1.00
		6000	99	98 *(94-101)*		56	68	1.21
	θ=10, X_0=K	4900	50	101	NA	46	603	13.19
		6000	99	98 *(94-101)*		56	655	11.69
	Y=120	4198	50	50	83-84 & 89-90	39	39	1.00
		7500	70	66 *(65-68)*		70	81	1.16
		40750	90	95		380	682	1.79
	Y=120, X_0=K	40750	90	95		380	1794	4.72
	Y^m = 4	7842	50	96	42-96	73	278	3.80
		12394	70	96		116	570	4.93
		55000	90	92 *(88-94)*		513	1296	2.52
K_b = 15		7842	50	37(9-47)	71-95	73	74	1.01
		12394	70	91		116	196	1.70
		55000	90	95		513	1071	2.09
K_b = 20		7842	50	95	53-94	73	128	1.74
		12394	70	92		116	493	4.26
		55000	90	96		513	1487	2.90

* Stock initialized, X_0, at %20 of K unless otherwise indicated. **NPR = Nepalese Rupees. ***Mean (minimum to maximum) ****Baseline parameters: Maximum pool of participants/households Y = 60; snare density s = 100/km^2; minimum participants for live-capture and release Y^m = 12; shape parameter θ = 1; intrinsic growth r = .19; discount rate δ = 12%; and adult male capacity $\overline{N}$ = 64.

6 DISCUSSION

Very few incentives currently exist at the local level to protect *in-situ* populations of musk deer across Nepal and other range states. The subsidization of conservation schemes that provide local communities with financial incentives could provide a means to improve the current situation for many musk deer populations. Two promising schemes are community-based wildlife management through the live-capture and release of musk deer and payment for ecosystem services through the direct financing of enforcement. Both have the propensity to increase musk deer populations. However, the relative benefits of PES or CWM depends on the resource condition and capture efficiency.

6.1 Comparison of Payment for Ecosystem Services and Community-based Wildlife Management Results

For the donor with a preference towards conservation outcomes, PES is preferable at lower densities of musk deer because harvesting is not economically viable. At low stock levels, there is a risk that a subsidy toward CWM could actually encourage a pulse control that reduces enforcement effort in order to harvest musk deer. Conversely, a CWM scheme has greater conservation and development potential at high stock levels, especially when the harvesting activities are able to cover operational costs. Provided favourable resource conditions, the need for an annual subsidy may be unnecessary because the revenue from harvesting musk is sufficient to: (1) encourage community members to increase enforcement effort in order to increase the musk deer stock and secure profit in future years, and (2) cover adequate enforcement costs.

A mix of both strategies may be encouraged when there is potential to harvest stock at financially sustainable populations in the *future*. In such a case, a PES strategy could drive stock levels up to profitable levels. At this point, the donor could switch from annual payments to a one-time start-up cost for CWM. However, if a donor desired additional conservation outcomes beyond the CWM profit maximizing stock levels, then funds through a PES scheme would be more cost effective to attract supplementary levels of enforcement. A preferable option would be just to pay for additional enforcement beyond that protected by the CWM.

6.2 Management Implications

My analysis does not infer specific results for any particular community because the parameters come from multiple sources. However, the modelling approach provides a framework to assess the potential conservation and development outcomes at a site for either strategy (especially if more detailed parameters can be collected at the local level).

When stocks are economically viable, then investing in a CWM strategy could save a donor long-term costs in enforcement and could provide substantial economic returns to the community. My model shows that the carrying capacity of adult males, $\overline{N}$, turns out to be an important parameter as it largely influences the optimal stock level for a CWM strategy that is economically self-sufficient in terms of operational costs. The equilibrium stock level at profit maximization is likely higher than many other wildlife harvesting schemes as the optimal stock size is not where the change in X is highest, but levels where N approaches $\overline{N}$ (assuming the X where $\overline{N}$ occurs is higher than max ΔX).

However, the carrying capacity, K_b, of musk deer is likely to be low in many areas of Nepal, and low K_b results in lower economic potential. Given the lower economic potential of these populations to cover operational costs and that the CWM strategy generally does not contribute to an increase in equilibrium at low stock levels, a donor would likely prefer a PES strategy for many regions of Nepal. In addition, a low α_v (high h_v) due to musk deer flushing at greater distances, could even make high values of K_b less lucrative for CWM (Harris and Guiquan 1993).

6.3 Limitations and Possible Extensions

One of the benefits of this research is the identification of data gaps and needs. The model developed in this paper made a number of assumptions in order to compare conservation strategies. Future data collection could help recalibrate the model so that relationships and parameters are more relevant to the resource problem in both general terms and for specific locations. Furthermore, there are potential extensions this model could include in order to incorporate greater levels of complexity.

The biological parameters r and θ were largely unknown, and took on large variations in the sensitivity analysis. Rates of growth can play a role in determining optimal enforcement levels and the viability of a CWM scheme. The biological parameters for wild populations of musk deer are lacking due to the paucity in long-term monitoring of *in-situ* populations (i.e., wild populations not farmed populations). Determining appropriate values for r would require tracking individuals within a population over time to collect age specific information on survival and fertility rates (Harris and Metzgar 1993). However, the solitary and cryptic nature of musk deer makes such detailed studies costly to implement.

In my model, the CWM harvest technique is a live capture and release strategy with the use of a drive net to capture animals. This technique was selected due to the availability of information from several studies that had successfully applied it in capturing musk deer, and because it is considered a low-cost, relatively harmless, and traditional method (Kattel and Alldredge 1991, Harris and Guiquan 1993, Wood et al. 2008). Other options for harvesting musk deer include the use of firearms, tranquilizers, and snares that hold the animal but do not kill it. Firearms would allow for sex selectivity and would require setting a sustainable quota. Additional information needs would include the impacts of adult male off-take on population growth, as a loss in males could affect female fecundity (Milner-Gulland 1997). The use of remotely delivered tranquilizers have a greater risk of injuring or killing musk deer than nets, as musk deer are small and inhabit areas with steep terrain. The use of no-kill snares has not been explored in much detail, however additional threats to the captured musk deer would include increased predator off-take between periods when snares are checked and possibly higher incidents of injury (per. com MJB Green 2008). Given additional information, these and other potential harvesting techniques could be included as extensions to the model.

Under the poaching dynamics presented in this paper, all snares were set at $t = 0$. Improvements to the poaching dynamics sub-model could provide additional reality to the real-world situation. Snares could enter the system at a slower rate rather than all set at an instant. Cox and Walters (2002) propose a modelling framework that incorporates fish that enter a vulnerable/non-vulnerable state for a recreational fishery. A similar approach could occur for snare entry (i.e., snares present/not present) in order to model their “removability” from anti-poaching efforts. However, an additional consideration under such a scenario is how a community would optimize the allocation of available participants, *I*, throughout the poaching period since the level of risk to the stock and the potential snare removal rate is more dynamic over time. The allocation of enforcement is optimal in my simplified model because the maximum number of available participants, *I*, is set at the start of the poaching period (i.e., when the poaching threat is the highest). Incorporating the rate of snare placement into the poaching dynamics remains a possible extension for future analysis.

Other model assumptions with respect to the poaching sub-model include (1) the use of snares as the dominant poaching technology and (2) the poacher as an outside agent (Also refer to section 4.2 for discussion). These choices reflect the situation in Phortse, Nepal. However, the use of firearms is also prominent poaching technique for killing musk deer in other areas. The presence of firearms may increase the costs for enforcement. For instance, the community may be less willing to apprehend poachers with firearms given the heightened risk to personal security. An explicit mention of firearms was not part of the original household survey conducted by Dr. Knowler's research team. As such, additional surveying may be required to develop a model that incorporates firearm hunting.

If local community members engage in poaching, then the modelling framework may require adjustments. In a CWM scenario, the allocation of property rights may provide community members with incentive to harvest populations sustainably. In a PES scenario, poachers could act as service providers and be paid not to poach (See Muller and Albers (2004)). Several papers have interviewed musk deer poachers and collected valuable data (Jackson 1979, Khan et al. 2006), but further surveys would likely be required to help identify the motives of poachers and their economic tradeoffs in order to identify appropriate PES and CWM schemes.

The current model allows for participants willing to participate at payment levels below Nepal's minimum wage (i.e., < 200 NPR). It is possible that members from a community could enjoy non-economic benefits from participating in either conservation strategy and would be willing to participate for other reasons. However, future analysis should consider adding a minimum wage as a constraint to take a more prudent approach. Interestingly, if community participation were open to free entry based on payment level, then a minimum wage constraint would result in a restriction to the minimum number of participants (i.e., similar to the harvest constraint Y^m , but applied to *both* CWM and PES strategies).

Current legal restrictions favour the implementation of PES. National and international laws prohibit the sale of musk from Nepal. Changes in legislation would be required if live-capture harvesting was to be a possible strategy within the country. Furthermore, a certification scheme would need to be developed and a sophisticated protocol to ensure that musk exports came from sustainably managed populations. This implies significant transaction costs on the CWM strategy. On the other hand, paying a community on the condition that snares are removed may be difficult to actually implement. How do you prove snares are initially present and are effectively removed? How do you avoid issues of community members who benefit from free riding (i.e., not actually participating in enforcement)? This paper developed a theoretical model based mostly on the comparison of community-level operational costs. However transaction costs also influence the relative efficiency of the two conservation strategies. The bulk of transaction costs for PES involve gathering information on environmental service providers, contract negotiation, and monitoring and enforcement (Neef and Thomas 2009). The cost of a monitoring program to assess musk deer population levels as part of a PES scheme, could be extremely expensive. However, similar monitoring costs may not be required for CWM. Future analysis should consider transaction costs as a component of the comparison. Other possible extensions include changing the payment scheme, such as paying community members for the number of newborn musk deer. This would allow the environmental service to match the conservation outcome in an even more direct fashion. However, careful consideration of the payment scheme would be required to match the payments with the costs of enforcement incurred by the community.

A risk analysis of parameter values would help refine future research endeavours. This REM 699 research paper includes a sensitivity analysis of parameters; however, a more detailed Monte Carlo simulation or similar approach, such as a Latin-hypercube sampling, would help to isolate key parameters that strongly influence the performance of CWM and PES strategies. Identifying significant parameters and reducing areas of uncertainty could help reduce costs of pilot projects, and help in the design of on-the-ground projects to test the feasibility of PES and CWM strategies.

7 CONCLUSION

Wildlife conservation will inevitably require context specific management strategies to deal with particular resource and social economic conditions. No single conservation approach will ever serve as blanket strategy that can apply to all situations. Both PES and CWM provide promising approaches for protecting wild populations of musk deer in developing countries as they provide incentives at the local level. Recipients, with a profit-maximizing objective, will always prefer a CWM approach as they receive a larger transfer of funds. The live-capture technique provides economic potential when resource conditions are good (i.e., higher stock levels) and/or when the technology is efficient at capturing animals. On the other hand, PES has the potential to induce larger enforcement effort at the margin of profit maximization and to serve as a more appropriate policy when stock sizes are too low. There is also the potential for a mix of both strategies to serve as the optimal policy. This would occur when PES can drive a population to a level that is economically viable to operate a CWM strategy.

Due to the paucity in data, and the uncertainty and complexities surrounding musk deer conservation, the refinement and comparison of PES and CWM schemes would benefit from *in-situ* pilot projects. Pilot projects that implemented either PES or CWM schemes at a small scale would allow resource managers to understand resource problems in more detail and help them to develop improved models of reality. Furthermore, with the information obtained from pilot projects, managers and donors could identify appropriate solutions towards the successful conservation of musk deer in the wild.

APPENDIX A – MODEL PARAMETERS, STATE AND CONTROL VARIABLES

Base case parameters, state variables and control variables are defined in Table 6.

Table 6. Model parameters, state variables and control variables

Symbol	Definition	Type	Value (Base Case)	Source
A	Size of total management area	Parameter	10 km^2	NA
c	Annual fixed costs of harvest management (i.e., nets, propane, supervision) NPR/year	Parameter	13,666	Wood et al. 2008
d	Number of hours poaching activities are active per year with maximum of 90 days/year.	State variable	NA	Derived. Max from per. comm. Green 2008.
E_t'	Harvesting effort by community (hrs)	Control variable	NA	Derived
E_t''	Anti-poaching effort by community(hrs)	Control variable	NA	Derived
E^m	Minimum total effort required for harvesting activities to occur in year *t* (hrs). $Y^m \cdot l \cdot 15$.	Parameter	1080 hrs (12 participants)	Kattel and Alldredge 1991
$\bar{E}$	Maximum total effort available by community, $Y \cdot l \cdot 15$	Parameter	5400 hrs (60 participants)	Wood et al. 2008
E^T	Total effort by community in year *t* (hrs), $E'+E''$	Control variable	NA	NA
f_m	Fraction of adult males in stock below $\bar{N}$ constraint	Parameter	0.17	Based on Jackson 1979
g	Total effort spent searching in year *t* per harvest group. A fxn of E'.	State variable	NA	NA
g''	Number of group hours spent on enforcement	State variable	NA	NA
H	Harvest in year t. Number of males captured.	State variable	NA	NA
h_τ	Handling time to remove snares $h_{\tau(random)}$= hours/s , $h_{\tau(barrier)}$= hours/$ş$	Parameter	$h_{\tau(random)}$ = 0.0166 hour (1 minute) $h_{\tau(barrier)}$ = 0.166 hour (10 minutes)	*per comm.* Green 2008
h_υ	Handling time of musk deer (hrs/N)	Parameter	1.5	Estimated from Kattel and Alldredge 1991, and Kattel 1992.
I_t	Individuals participating in CWM or PES project in year *t*	State variable	NA	NA
K	Carrying capacity of management area (X)	Parameter	500	NA

K_b	Maximum total musk deer density (X/km^2)	Parameter	50	Kattel 1992
l	Labour provided by one participant (hours/year)	Parameter	90	NA
n	Territory of adult male musk deer (km^2)	Parameter	0.155	Kattel 1992
$N(t),N_t$	Number of adult males at time t. *$N(t)$: instantaneous, N_t: discrete*	State variable	NA	NA
$\overline{N}$	Carrying capacity of adult males due to territoriality (A/n)	Parameter	64.5	Based on Kattel 1992
P	Poaching Deaths	State variable	NA	NA
p	Price of per harvested musk deer in Nepalese Rupees community receives.	Parameter	19350	Based on NPR 1500/g (Wood et al. 2008) and derived with average musk produced per deer (12.9g/deer). (Weigo and Shuyan 1991)
r	Intrinsic rate of growth	Parameter	.194	Oli and Zinner 2001
$s(0)$	Number of snares set at the start of each year	Parameter	100/km^2*A	Green 1986.
$s(t)$	Number of snares	State Variable	NA	NA
$\bar{s}$	Average number of snares discovered & removed once brush barrier is detected.	Parameter	10	NA
T	Number of years to run simulation.	Parameter	80	NA
V	Maximum net present value in period t	State variable	NA	NA
X	Total musk deer stock	State variable	NA	NA
$\widetilde{X}$	Musk deer stock that survive poaching activities prior to population growth	State variable	NA	NA
$\bar{x}_2$	Mean value for wealth status. Used in probit model.	Parameter	0.384	Data provided by D. Knowler (unpublished)
$\bar{x}_3$	Mean value for Group 3 member. Used in probit model.	Parameter	0.316	Data provided by D. Knowler (unpublished)
Y^m	Minimum number of participants required for harvesting to occur in year t.	Parameter	12	Kattel and Alldredge 1991
Y	Total number of households in village, maximum number of participants	Parameter	60	Wood et al. 2008
z	Annual Subsidy (Nepalese Rupees). Parameter z set to approximate stock size of 50%, 70% and 90% of K under PES strategy.	Parameter	8210, 12500, 74000	Derived
α	Instantaneous death rate per snare, where $\alpha_{(random)}$ = random placement, $\alpha_{(barrier)}$ = placement along fence barriers (/hr•s)	Parameter	$\alpha_{(random)} = 4.42e^{-07}$ $\alpha_{(barrier)} = 4.63e^{-06}$	Derived
a_τ	Search rate to remove snares (km^2/hr)	Parameter	$a_{\tau(random)} = 0.00134$ $a_{\tau(barrier)} = 0.01344$	Derived
a_υ	Effective search rate for musk deer CWM harvest (/hr)	Parameter	0.003	Estimated from Kattel 1992, Green 1985,

				Focardi et al. 2005, and Gill et al. 1997.
β_o	Intercept of probit model	Parameter	-4.489	Data provided by D. Knowler (unpublished)
β_1	Coefficient of probit model for payment	Parameter	0.0005	Data provided by D. Knowler (unpublished)
β_2	Coefficient of probit model for wealth status	Parameter	7.141	Data provided by D. Knowler (unpublished)
β_3	Coefficient of probit model for group 3 member	Parameter	-1.584	Data provided by D. Knowler (unpublished)
δ	Discount rate	Parameter	12%	Wood et al. 2008
θ	Shape parameter for carrying capacity dependence	Parameter	1	NA
π	Profit per individual	State variable	NA	NA
π^T	Total profit for community	State variable	NA	NA
ρ	Discount factor = $1/1+\delta$	Parameter	.893	NA
Φ	Cumulative proportion of households willing to participate based on probit model	State variable	NA	NA

APPENDIX B - PARAMETER ESTIMATIONS

Community Harvest Parameters

The derivation of handling time, h_υ, and instantaneous capture rate, α_υ, parameters for equation 4 are provided below.

Handling time

Handling time, h_υ, was derived from the following equation:

$$h_\upsilon = \frac{T_y}{\rho_s} + T_z \qquad \text{[A1]}$$

where T_y is the time spent pursuing a target animal, T_z is the time spent extracting musk once the animal is caught, and ρ_s is the proportion of animals successfully caught (where $0 \leq \rho_s \leq 1$). To solve for h_υ, parameters on the right-hand side of Equation A1 are set as: T_y= 0.42 hours to set up nets and make drive attempt per animal; T_z= 0.75 hours for animal to recover from sedation; and ρ_s = 0.56 as the proportion of captures successful (Kattel and Alldredge 1991, Kattel 1992). When solved under the above conditions h_υ= 1.55 hours/deer captured.

Search rate for Musk deer

The effective instantaneous search rate for capturing musk deer 'α_υ' can be described as:

$$\alpha_\upsilon = (2W_\upsilon V_\upsilon \rho_s)/A \qquad [A2]$$

where W_υ is the width of search path, V_υ is the relative velocity, ρ_s is the probability of successful capture, A is the size of the management area (adapted from (Holling 1966, Gendron 1984)). If t is considered infinitesimal then α_υ is essentially unitless. Estimating the a_υ parameter is more difficult since not a lot of data is available. An initial attempt to quantify a_υ makes a number of assumptions. Relative velocity, V_υ, was simplified by assuming musk deer do not move during the search, which is a reasonable assumption as musk deer often rest mid-day when capture efforts are most likely to occur (Kattel and Alldredge 1991, Focardi et al. 2005). No data was available for the speed of a harvest group. Thus the use of a proxy is based on Focardi's (2005) census study of Roe Deer, where the average walking speed of observers through an oak forest was recorded at 1.34 km/hr (note: observers were to walk at a speed that would not flush the animals). Thus V_υ = 1.34 km/hr. The width of search path, W_υ was set at 0.02 km based on observations from Green (1985). He records that visibility greatly diminished beyond 20 m in forested areas of his study site. Similarly, Gill et al. (1997) estimate the mean visibility for observing deer lying down and standing up in thickets of four forest areas located in the UK, is 20.2 m and 25.3 m respectively.

Poaching Parameters

Instantaneous death rate, $\alpha_{barrier}$

A random walk model (RWM) was developed to find $\alpha_{barrier}$. The RWM model is a simplification of the true poaching scenario, but is likely an improvement to the assumption that snares are randomly placed throughout musk deer habitat. In the RWM snares are set along the borders of an individual musk deer's home range. Since a musk deer's home range, *R*, is between 15-35 hectares the number of snares are adjusted from s/km^2 to s/R for different values of *R* per simulation. The total area for *R* is spatially set up as a homogeneous square area composed of a finite number of cells. Each cell has an area 0.0001 km^2, which is based on the length and height of a typical snare. Musk deer movement occurs once in each hourly time-step and is assumed to represent a musk deer browsing shrubs scattered uniformly within the forest. The mean distance travelled per hour, *v*, is derived from Green (1985), who followed the movements of a musk deer travelling for a 22 hour period (Note: Sample Size =1). An hourly update to the musk deer's position is solved by:

$$B_t = 2\pi U \qquad [A3]$$

$$x_t = \cos(B_t)\cdot v\cdot\varepsilon,\ \varepsilon\sim N(0,1) \qquad [A4]$$

$$y_t = \sin(B_t)\cdot v\cdot\varepsilon,\ \varepsilon\sim N(0,1) \qquad [A5]$$

where *t* , in equations A3,A4 and A5, is a subscript for hourly intervals, B_t is a polar coordinate for the direction of musk deer, *U* is a number between 0 and 1 randomly generated from a uniform distribution, π is a constant (i.e., 3.14…), x_t and y_t are the directions moved along the horizontal and vertical axis respectively, v is the mean distance per hour a musk deer travels, and ε is an error term with a standard normal distribution. When the musk deer confronts a fence barrier which is assumed to run along the border of the homerange, then there is a 50 percent chance the musk deer will walk along the barrier as the musk deer does not change its 'intended' turning behaviour but is inhibited to cross the barrier (i.e., its' x or y direction is set to zero and only the other direction is updated). If the deer walks along the fence, then a count is made of each time it passes a snare trap within a discrete step. For each snare trap that the musk deer passes I assume the individual has a ¼ probability to pass through the trap.

The RWM ran 1000 times per fixed snare density in a Monte Carlo simulation (Snare densities, s, were set at 25 unit intervals per km^2 up to 600 snares). Each run would finish when a snare trapped a musk deer. Assuming the risk of death to a musk deer during the poaching season is constant over time, the hazard function takes on an exponential distribution and an estimate of the instantaneous hazard rate, h_z, for a particular density of snare traps is given by the inverse of mean time until death (Crawley 2007). That is,

$$h_z = 1/\mu$$

where μ is mean hours until death occurs.

Figure 14A provides a histogram of time until death for 1000 iterations when snares were set at $400/km^2$. The line passing through the histogram is the density function, solved as:

$$f(t) = h_z e^{-h_z t}$$

and provides an approximation of the number dying around time t. This model assumes that the number dying at each time-step declines exponentially.

Finally, the parameter $\alpha_{barrier}$ was estimated as the slope of the linear regression between h_z and the density of snare traps (Figure 14B). The estimate for $\alpha_{barrier}$ is used in the main simulation in order to model poaching deaths from fence barriers.

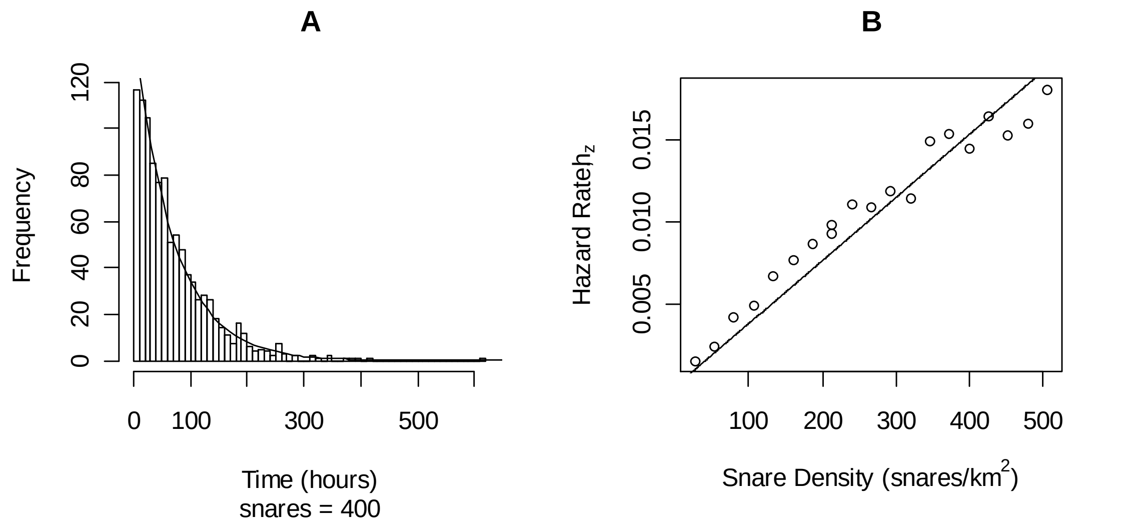

Figure 14. Sample of results from random walk model for estimating death rate per snare, $\alpha_{barrier}$. Panel A -An example histogram of deaths overtime from the random walk model, iterations = 1000, snare density = 400 snares/km^2. Note: the instantaneous hazard rate, h_z, is calculated as 1/μ (mean time until death). The line through the histogram is calculated with equation (f(t)) as shown in Panel A. Panel B – Displays snare density , s/km^2, vs hazard rate h_z . Points are results from random walk model. The line is a linear approximation of the relationship to estimate $\alpha_{barrier}$.

Table 7. Additional parameters used for parameter estimation

Symbol	Definition	Type	Value (Base Case)	Source
B	Polar coordinate derived in RWM	State variable	NA	NA
D	Distance required between musk deer and a snare trap in order to trigger a trap (km) – See Appendix A	Parameter	0.00005km (5cm)	Rowcliffe et al. 2003
h_z	Hazard Function	State variable	NA	NA
T_y	Time spent pursuing target animals once detected for CWM. See Appendix A	Parameter	0.42	Kattel 1992
T_z	Time spent extracting musk once musk deer are caught. See Appendix A	Parameter	0.75	Kattel 1992
v	Average velocity of musk deer (km/hr). See Appendix A	Parameter	.044 km/hr (1.06 km/day)	Derived from Green 1985
v_υ	Speed of community harvest group. See Appendix A	Parameter	1.34 km/hr	Derived from Focardi 2005
W_υ	Width of search path for community harvest group searching for musk deer. See Appendix A	Parameter	0.02 km	Derived from Green 1985
μ	Mean hours until death occurs from snaring. Value derived from RWM	State variable	NA	NA
ρ_s	Proportion pursuits of musk deer that lead to a successful capture	Parameter	.56	Kattel and Alldredge 1992

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