Economics of wildlife management—an overview

Ing-Marie Gren¹ · Tobias Häggmark-Svensson¹ · Katarina Elofsson¹ · Marc Engelmann¹

Received: 18 November 2016 / Revised: 21 February 2018 / Accepted: 13 March 2018 / Published online: 27 March 2018
© The Author(s) 2018

Abstract

This study makes an explorative overview on two main research topics in economics of wildlife management: determination of population sizes and policy design. The results point out a large and comprehensive research on each of these issues, in particular on the estimation of values and costs of wildlife, where this information is necessary for the determination of population size. A drawback is that most of the value and cost studies do not relate their estimates to wildlife population size, which limits their usability for efficient policy design. Most valuation studies estimate the recreational value of hunting, which can range between 13 and 545 USD/hunting day (in 2013 prices), and two thirds of the included studies have been applied to wildlife in the USA. A majority of the studies on the costs of wildlife management calculate losses from carnivore predation on livestock and ungulate damage to crops, while a few consider dispersal of diseases and the cost of traffic collisions. Unlike valuation studies, several of the cost estimates apply to wildlife in developing and emerging economies. With respect to policy design the literature, which is mainly theoretical, suggests economic incentives for conflict resolution, where those suffering from wildlife damages are compensated for their losses. However, there are some issues which remain to be addressed by economists: relating costs and benefits to wildlife populations; estimating values and costs of wildlife in developing countries; evaluating wildlife policies in practice; addressing implications of uncertainty in population size, costs, and benefits for policy design; and estimating transaction costs for implementation and enforcement of wildlife policies.

Keywords Costs · Benefits · Policies · Economics · Wildlife · Review

Introduction

Wildlife is a source of both costs and benefits to society. Costs occur from wildlife predation on livestock, destruction of crops, traffic collisions, and transmission of diseases to animals and humans. Benefits accrue from hunting, recreational activities, food, and other ecosystem services. The asymmetric allocation

of these costs and benefits among stakeholders constitutes a threat to wildlife. Approximately 50% of all mammals worldwide are in decline, and 25% are facing extinction because of illegal hunting (Treves and Karanth 2003; Pack et al. 2013) and destruction of habitats, e.g., for agricultural purposes (Roemer and Forrest 1996; Woodroffe and Ginsberg 1998). On the other hand, economic losses due to carnivore predation on livestock in eastern and southern Africa can range from 1 to 25% of the potential revenue, and carnivores can severely reduce the quality of life (Bulte and Rondeau 2007). Increased land use for agricultural purposes, which is occurring mainly in developing countries, can raise this type of cost (Madhusudan 2003). However, wildlife also provides a source of recreational benefits and income, where hunting and animal viewing can generate significant profits (Barnes et al. 1999; Hofer 2002).

In principle, the conflicts among stakeholders and threats to wildlife could be mitigated by implementation of policies which adjust for the asymmetric allocation of costs and benefits and create incentives for wildlife preservation and restoration. When some stakeholders are not fully compensated for losses due to wildlife while at the same time other stakeholders obtain

✉ Ing-Marie Gren
Ing-Marie.Gren@slu.se

Tobias Häggmark-Svensson
Tobias.Haggmark@slu.se

Katarina Elofsson
Katarina.Elofsson@slu.se

Marc Engelmann
Marc.Engelmann@slu.se

¹ Department of Economics, Swedish University of Agricultural Sciences, Uppsala, Sweden

revenues generated by wildlife, there exists a market failure which requires policy intervention (e.g., Tietenberg 1997). However, despite the concern about wildlife management since the 1950s (see, e.g., Gordon et al. 2004), there has been no survey of studies in economics of wildlife management. In principle, economics of managing any renewable resource raises two main questions: (i) what is the optimal size of the populations, i.e., the population size that maximizes net benefits for society, and (ii) if actual populations deviate much from the optimal level, how to change human behavior to reach the optimal population levels (e.g., Tietenberg 1997). The first question is generally approached by bio-economic modeling or benefit-cost analysis. The former allows for determination of the optimal population sizes, while the latter usually calculates costs and benefits for specific projects not necessarily implying optimal population levels. The second inquiry is addressed by applying methods from economics of regulation. These two research fields have a long tradition in economics.

The purpose of this study is to investigate how the questions on optimal population size and policy design have been formulated and approached in wildlife economics. In order to carry out the overview, we need to define what we mean by wildlife. Traditionally, wildlife refers to non-domesticated wild animals that could be hunted for recreational or food provision purposes (Yarrow 2009). More recently, the definition includes all non-domesticated and non-cultivated animals and plants. In practice, many wildlife management organizations and public authorities define wildlife as land-based mammals and birds. Most studies on wildlife management focus on these species, which could be indigenous or exotic to the region or habitat under study (Yarrow 2009). In the present overview, we narrow the definition to include land-based animals and birds that can be hunted, which is close to the early definition of wildlife. Most economic studies which explicitly address wildlife management apply this definition. Our definition of wildlife then excludes the large body of literature on the economics of biodiversity (see Loomis and White 1996 for a review) and on the economics of invasive species (see Gren 2008 and Marbuah et al. 2014 for reviews). Threatened and invasive species undoubtedly belong to and affect wildlife, but the focus of the existing studies is mainly on costs and benefits of preservation or mitigation, respectively, and not on the balancing of the costs and benefits generated by a particular species, and policies that support this.

There are a few previous reviews of economic studies on wildlife management (Shwiff et al. 2013) and comparisons of policy regimes (Rasker et al. 1992; Mawdsley et al. 2009; Pack et al. 2013). Shwiff et al. (2013) provide a comprehensive and critical review of the methods used to calculate costs and benefits of wildlife conservation projects. The authors emphasize the difficulty of calculating benefits from wildlife projects because of the existence of non-market values, although measures have been developed to assess these benefits

over the last 50 years. Relative merits of different evaluation methods without the use of benefit estimates (cost-effectiveness analysis, cost-utility analysis, threat reduction assessment, and conservation output production years) are also discussed. Moreover, it is pointed out that secondary effects of wildlife projects, i.e., effects beyond the direct primary costs and benefits, can be significant. However, regional economic models capturing these effects have been applied mainly to regions in North America and Europe, while applications to other regions remain to be performed.

Rasker et al. (1992) provide a general discussion on the merits of economic incentives in wildlife management and make a brief comparison of policies suggested in the literature and policies used in practice. They claim that economic studies usually compare the polar cases of a “free market” or commercialization of wildlife, where all hunting rights and their uses are exchanged on a market, and public ownership of wildlife. A common conclusion is then that privatization of wildlife would create incentives for conservation. Rasker et al. (1992) challenge this conclusion by discussing different types of market failure which erode the potential of free market solutions, and point out the advantages provided by mixed private and public ownership used in practice in many countries. This is supported by Pack et al. (2013), who review and compare studies on actual policies in different parts of the world. They define policy regimes along two dimensions: wildlife ownership (private or public) and source of conservation funding (public taxes or fees from, e.g., hunting and eco-tourism). They classify six countries (USA, Canada, South-Africa, Tanzania, Kenya, and India) into different combinations of these dimensions and compare the performance of the policy regimes with respect to wildlife populations, local community funding, benefit/cost ratios of wildlife, and protected areas. They show that the so-called North American model, with public ownership of wildlife and a combination of funding from public taxes and hunting fees, performs best according to these criteria of comparison.

Another form of policy-oriented study is carried out by Mawdsley et al. (2009), who reviewed selected strategies by countries (USA, Canada, England, Mexico, and South Africa) for mitigating climate change impacts on wildlife. These strategies include identification of necessary measures, such as land and water conservation and species translocation, and how to implement these measures, i.e., which policies to use. According to Mawdsley et al. (2009), the necessary policy tools exist in current management, but need to be enforced with more stringency in order to meet the climate change threats to wildlife.

In our view, the main contribution of the present study is the extended scope compared with other review studies where we include studies on calculations of costs and benefits of wildlife, efficient wildlife population levels, and policy design. This is complementary to the review of calculation

methods made by Shwiff et al. (2013). It also updates the findings by Rasker et al. (1992) on the economics of policy design for wildlife management. A limitation of the present study is that we do not attempt to explain or critically evaluate differences in studies in these fields, which would, e.g., require meta-analysis of existing cost and benefit studies. Instead, we aim to identify main approaches and results on the two main issues on efficient game population sizes and policy design raised in economics, which wildlife species have been at focus, and eventual gaps in the literature.

A systematic search of studies was made for relevant literature in scientific journals and in the “gray” literature, which includes reports from authorities and consultancy firms. Publications were found by entering the key words “wildlife,” “game,” and “economics” together with either “efficiency,” “benefits,” “costs,” or “policies” in Web of Science, Scopus, and Google Scholar. No restriction was made with respect to year of publication and database. This was complemented by a snowball method, where we follow the references in and citations to all relevant studies in the mentioned databases. In total, we found and included 102 studies addressing the first question on population size and 32 studies focusing on the second question regarding policy design. The total number of studies, which amounts to 131, is slightly lower than the sum of studies in each category since a few studies addressed both questions. We do not claim to present an exhaustive survey in any of the four categories, but hope to capture the main findings in all these categories. All costs and benefits reported in the literature are here expressed in USD in 2013 prices using the US consumer price index (CPI). Only a few studies express results in other currencies. We then convert these estimates into 2013 prices by using the respective country’s CPI and apply the respective exchange rate for 2013.

The remainder of this paper is organized around the two main questions raised in economics on game animals. The first question on the size of populations is examined in the “[Population size\(s\) of game animals](#)” section, which contains reviews of studies on benefits and costs of game animals, and calculations of optimal population sizes. The second question is on policy design which is addressed in the “[Wildlife policies](#)” section, where we survey studies suggesting and comparing different policies for improved wildlife management. The paper ends with some concluding remarks.

Population size(s) of game animals

In economics, the question on the size of a population is determined by its associated benefits and costs to society. It is then usually not enough to compare benefits and costs of a particular population size, but instead find the size which maximizes total net benefits to society the so-called optimal

population size. There could be many levels with a net surplus to society, but only one which maximizes this surplus. However, to calculate surplus and maximum surplus, there is a need for data on benefits and costs of game animals, and we therefore review studies on these topics in this section. A number of studies also attempt to calculate population sizes which maximize net benefits. Most studies focus on either benefits (47), or costs (42), or optimal population sizes (12).

Benefits of game animals

Game animals give rise to values for individual hunters and to society in general. The values attributed to individuals usually include both use and non-use values. The use values capture values associated with active use of the resource, such as meat value, hunting, and viewing, while non-use values include all other values outside the use category such as values assigned to the existence of a game animal (e.g., Desvousges et al. 1983; Boyle and Bishop 1987). In general, it is straightforward to estimate individuals’ value of meat of game by using associated market prices. Other values, such as recreational, viewing, and non-use values, which are not traded in markets can be more difficult to measure. Starting in the 1960s, there is a large body of literature on the estimation of such values (see, e.g., Tietenberg 1997). In principle, two main methods are suggested, which rest on revealed or stated preferences. The revealed preference methods rely on behavior in indirect markets when the condition of weak complementarity is fulfilled (Mäler 1974). When this is the case, there is a link between the market and non-market good, such as markets for travel and equipment for hunters which can be related to their valuation of hunting. These methods rest on a long tradition in the development of travel cost methods (TCMs), and hedonic methods where the value of wildlife game can be related to, for example, market price of land. These methods have limitations with respect to the estimation of non-use values.

In order to deal with the limitations of revealed preference methods, stated preference methods have been commonly applied, which derive use and non-use values from people’s stated willingness to pay (WTP) for hypothetical changes in the supply of a public good. The idea of using artificial markets was first suggested by Ciriacy-Wantrup (1947), who was the first to use interviews with hunters and recreationists to reveal their WTP for a specified recreational area in Maine (Mitchell and Carson 1989). Since then, stated preference methods in terms of contingent valuation methods (CVMs) have been vastly applied. Difficulties with these methods are that the stated WTP can be affected by the framing of the decision context and formulation of questions which is discussed in Shwiff et al. (2013). It can also be difficult to relate values

to the presence of a game. For example, Shaw (1984) points out that values of, e.g., camping in a forest are likely to be positive even without the presence of wildlife. More recent developments, so-called conjoint analyses or choice experiments, have been made to mitigate these problems by assessing the value contribution of different attributes of a public wildlife good, such as number of game and size of hunting land in studies applied on the valuation of game (e.g., Delibes-Mateos et al. 2014).

Studies estimating benefits to society, which we denote sector studies, calculate the role of the hunting sector for an entire regional or national economy. The simplest way of measuring the size of this sector is to record and aggregate firms' sales of equipment, transport, and lodging for hunting purposes (e.g., Lindsey et al. 2006). The expenditures will give rise to second-order effects in a region, where sectors with immediate benefits from hunting activities impact other sectors through their demand for deliveries. In order to capture all effects of hunting activities, social accounting matrices are constructed and input-output models are employed for the evaluation of total economic impact in terms of economic multipliers which show the dispersal effect of hunting activities in a region.

In this review, we included 48 valuation studies published during the period 1974 to 2017. The majority of these studies, 39, estimated individuals' hunting value of game animals, and the remaining 9 studies calculate the economic impact of the hunting sector (Table 1).

The two methods for valuation of WTP for game animals are applied in relatively equal proportions, where 18 studies used revealed preference methods and 21 stated preference methods. A majority (80%) of the studies on moose apply revealed stated preference methods, and the two methods are used in similar proportions for valuing other game. On the other hand, the valuation studies show high asymmetry with respect to choice of game animal, regional application, and, for hunting value studies, choice of value construct. Most of the studies calculate benefits of a single species, in particular moose and deer, which account for 67% of all studies included in this review. Approximately 70% of all studies estimate value of game animals in USA. With respect to the studies on hunting value of game animals, the most commonly applied value construct is value/day but the estimated level show differences within and between the species categories. The largest variation appears for deer, the value of which ranges between 7 and 545 USD/day. However, the highest value, which applies for deer hunting in the Scottish Highlands (Bullock et al. 1998), is an outlier. The values cited in the other studies range between 7 and 267 USD/day, which is in the same order of magnitude as for other game. A few studies used stated preference methods to estimate impacts of different attributes, such as abundance of animals, trophy hunting, and certainty in bagging, on the estimated value (Livengood

1983; Loomis et al. 1988; Morton et al. 1995; Fried et al. 1995; Bullock et al. 1998; Delibes-Mateos et al. 2014).

With respect to sector studies, most of them calculate expenses for hunting of all game animals without consideration of dispersal into other sectors in the economy. In countries where hunting is an important source of incomes from tourism, the share of gross domestic product (GDP) can amount to 7% (Lindsey et al. 2006). The two studies considering dispersal effects in regions in USA show that these can be at least 65% of the direct expenditures in the hunting sector (Burger et al. 1999; Grado et al. 2007).

The brief review in this section reveals the concentration of valuation studies on game animal in particular in USA. Studies on other game and in other parts of the world remain to be made. Studies calculating the role of the hunting sector for an economy, in particular on the economy-wide dispersal effects, are relatively few and recent which also remain to be made for countries outside USA. The methods as such are well anchored in the scientific community. There has been a slight shift in methods estimating only levels of values to stated preference methods which allow for estimating the role of different hunting attributes. A challenge for policy making is the lack of studies relating estimated values to number of animals, which is needed when determining optimal population sizes.

Costs of wildlife

Costs of game animals include browsing, predation, traffic accidents, and transmission of diseases. Similar to benefit estimates, these costs are calculated for groups of individuals with and without dispersal effects into the rest of an economy. For each category, the costs consist of three parts: costs for actual damage, costs of mitigation, and costs for adaption measures (e.g., Conover et al. 1995). For example, costs of ungulates for farmers include the actual damage to, e.g., lost yield and livestock, expenses for mitigation measures such as fencing of agricultural land, and adjustment by avoiding cultivation of crops preferred by the animals. Predation may also create indirect costs from reduced weaning weight, decreased conception rate, reduced weight gain, and increased livestock sickness (e.g., Rashford et al. 2010).

The literature has developed and applied two main methods for calculation of direct costs of browsing and predation: questionnaires to stakeholders on actual costs and analysis of compensation payments. A difficulty with survey methods is that farmers may have incentives to exaggerate the economic impact if they believe the responses affect compensation payments (e.g., Rollins et al. 1996). Actual compensation payments, which often rest on market prices of the game animal (e.g., Baker et al. 2008), may not reflect the true cost for the farmers such as outlays for preventive measures taken to protect them against predation.

Table 1 Classification of 48 studies estimating economic effects of hunting in different categories with respect to value measurement and animal under study, in 2013 USD (number of studies in parentheses)

Type of value estimate	Moose (14) ^a	Deer (13) ^b	Birds (7) ^c	Other animals or several game (14) ^d
Individual based studies (39 of which 18 RP ^e and 21 SP ^f)				
Value/hunting day (23)	41–207 (10)	7–545 (8) ^g	97 (1)	45–175 (4) ^h
Value/hunting trip (4)	129 (2)		30 (1)	255 (1) ⁱ
Value/animal (9)	247,455 (2)	12 ^j , 71, 90 ^k (3)	17 ^l , 66 (2)	29–324 ^{m,n} (2)
Value/year (2)		90 (1)	255 (1)	
Value/hunting season (1)			612 (1)	
Sector studies (9)				
Single hunting sector, billion (7)				0.03–35 (or 0.01–7% of GDP) ^o (7) ^p
Economy-wide impacts, billion (2)		0.9–1.3 (1)	0.3 (1)	

Main source: review in Häggmark-Svensson et al. (2015)

^a 9 in USA, 2 in Canada, 3 in Europe

^b 9 in USA, 1 in Canada, 1 in Europe, 2 in New Zealand

^c 5 in USA, 1 in Australia; 1 in Europe

^d 8 in USA, 2 in Europe, 4 in Africa

^e RP = Revealed preference method

^f SP = stated preference method

^g Schwabe et al. (2001); the seven other studies are found in Häggmark-Svensson et al. (2015)

^h 2 studies on upland game (rabbit, pheasant, quail, grouse, wild turkey, and dove), 1 each on all and big game

ⁱ All game

^j The value of an increase in the probability of bagging a deer by 1%

^k Keith and Lyon (1985)

^l Brown and Hammack (1973)

^m Moose (29 USD) and deer (50 USD) in Loomis et al. (1989)

ⁿ Wild boar (290 USD) and fallow deer (324 USD) in Mensah and Elofsson (2017)

^o Own calculations of ratios in Hofer (2002) and Lindsey et al. (2006) with the lowest percent in Spain and Angola and highest for the tourist sector in Namibia; GDP = gross domestic product

^p 1 study on leopards, 1 on bear, and 5 of game in general

Animal vehicle collisions with game animals create costs in terms of fatalities, personal injuries, and repair of vehicle damages. The costs of fatalities and personal injuries include valuation of life and health, the calculation of which has a long tradition in economics (e.g., Mishan 1971). These costs are considerably higher than vehicle repair costs. The calculation of expected cost of an accident with a particular game animal are then calculated based on probabilities and costs of these damages. Similar methods are used for estimating costs for transmission of diseases affecting humans, such as transmission of tick born encephalitis by roe deer and transmission of rabies in many countries (e.g., Conover et al. 1995). Effects on livestock are calculated as the value of lost animals.

In total, this review includes 43 studies on cost of game animals, half of which are applied to costs of livestock predation (Table 2).

Almost all studies (95%) on costs of livestock predation and crop damage apply market prices to elicit damage cost,

and the remaining studies use surveys and actual compensation payment paid to the affected farmers. Compared with valuation studies, the studies on the costs of wildlife are more dispersed with respect to regional applications; 51% of the studies refer to the USA, while the others are equally divided among countries in Europe, Asia, and Africa. Half of the studies on livestock damage and all the studies calculating costs of big cats are applied to developing or emerging economies. One reason for this can be the relatively recent focus on predation in these countries around national parks or animal refuges (Mishra 1997; Butler 2000; Rao et al. 2002; Madhusudan 2003; Ikeda 2004; Michalski et al. 2006; Gusset et al. 2009; Tamang and Baral 2008; Abay et al. 2011). A few of the studies on costs of livestock predation estimate both direct and indirect costs and show that the indirect cost can be of similar magnitude of order as the direct cost (Howery and DeLiberto 2004; Laporte et al. 2010; Steele et al. 2013; Ramler et al. 2014).

Table 2 Estimated costs of wildlife in 43 studies, in USD/year in 2013 prices (number of studies in parenthesis)

Type of cost	Dog species (9) ^c	Cat species (6) ^d	Carnivores ^a (7) ^e	Deer (5) ^f	Wild boar (4) ^g	Others ^b (12) ^h
Livestock damage (22)						
USD/household (20)	50–9,200 (8)	2–515 (6)	89–3,900 (5)			5 (1)
Other construct (2)			0.06% of agricultural value added, 20% decrease in agricultural profits			
Crop/forest damage (11)						
USD/household (9)				284–2,630 (2)	30–46 (3)	144–2,038 (4)
Other construct (2)						2, 450/acre; 383/animal
Others (7)						
Vehicle collision (5), USD/capita				3–5 (2)	1.5 (1)	30 ⁱ , 21 ^j (2)
Disease, million (2)						665 ^k , 695 ^l
Sector studies (3)						
Direct cost, million (1)						45,000 ^m
Dispersal effects, million (2)	3.1–4.5 ⁿ					37 ^o

Main source: review in Häggmark-Svensson et al. (2015)

^a Several carnivore species or carnivore in general without specification

^b Bear, geese, elephants, and hyena

^c 4 USA, 1 middle east, 2 Europe, 2 Africa

^d 2 Africa, 4 Asia

^e 5 USA, 1 Europe, 1 Asia

^f 5 USA

^g 6 USA, 1 Canada, 3 Europe, 2 Asia

^h Traffic accidents with deer and moose in USA (Huizer et al. 2008)

ⁱ Traffic accidents with moose, roe deer, and wild boar in Sweden (Gren and Jägerbrand 2017)

^k Costs of rabies on human and livestock health in Asia and Africa

^l Wildlife disease in general in USA

^m Costs to farmers, foresters, and of traffic accidents and disease transmission of all wildlife in USA (Conover et al. 1995)

ⁿ Regional effects of coyotes in UT, USA, and 70% of total cost are dispersal effect (Taylor et al. 1979)

^o Direct costs and dispersal effects of coyotes, dogs, and mountain lions in USA (Jones 2004)

With respect to studies on other costs, deer and wild boar have been subject to a majority of the studies on costs of crop/forest damage and traffic collisions, which are mainly applied to USA. Crop damage by other animals includes damage by black bear and geese in the USA, elephants in India, and European bison in Poland. Similarly to livestock predation, damage to crops can occur to farmers close to or inside national parks or animal refuges. The few studies on costs of transmission of diseases can also be noticed.

Similar to benefit studies, there is a consensus in the scientific community with respect to quantitative methods for estimating costs. Although studies on costs of wildlife show more regional dispersion than benefit studies, there is still a concentration to USA, where slightly more than 50% of the studies have been applied. Applications to other regions remain to be made also for cost estimates. Similarly, the scope of application needs to be enlarged to include studies on costs of vehicle

collisions and transmission on diseases. Studies on cost estimates perform worse than benefit studies with respect to relating cost construct to population size, where none of the studies in this review presented costs related to animals or population sizes.

Efficient wildlife population(s)

Most of the studies reported in the “Benefits of game animals” and “Costs of wildlife” sections estimate either benefits or costs from game animals. Only a few studies compare cost and benefits to examine whether the population size is too high or low. Horne and Petäjistö (2003) found that the bag value per moose animal is higher than the costs of browsing for land owners in Finland. However, they cannot make any conclusions whether the population should change and if so by how much. An

economically appropriate evaluation of optimal population sizes should build on marginal costs and benefits. If a marginal increase in the population implies that benefits increase more than the cost, the population should increase. On the other hand, if a marginal decrease in the population generates higher cost than benefit increase, the population should decrease. This is the issue addressed by studies examining optimal population sizes which maximize benefits minus costs for society. To do this, there is a need for quantitative information on how costs and benefits change with population sizes. However, such data is not available in any of the studies reviewed in the “[Benefits of game animals](#)” and “[Costs of wildlife](#)” sections. Furthermore, it is necessary to have data on population dynamics which show how populations grow and are affected by environmental conditions and different kind of human pressures such as hunting and vehicle accidents.

Studies calculating optimal wildlife populations rest on advances in fishery economics (e.g., Clark 1990; Reed 1980). Estimates of actual populations and assumptions about population growth functions are fundamental basis of the analysis. Similar to fishery economics, there is a variety of methods for estimating wildlife population which include aerial surveys, mortality counts, and reports from hunters. Irrespective of method, the population estimate is uncertain.

Common designs of the population growth functions are logistic or stage structured models. The logistic model implies an *S*-shaped growth function with relatively rapid growth rate at low population levels and low growth rate at higher levels. This approach does not allow for selective hunting, i.e., targeting of specific classes of individuals, which is an advantage of the stage class models, where overall population growth depends on the number of individuals in different stages, such as yearlings and male and female adults.

The studies on optimal population sizes differ, not only with respect to choice of population growth function but also regarding choice of region, animal, and included value and cost items (see Table 3).

With respect to choice of population growth function, most studies apply the simple logistic model. This is made by Brown and Hammack (1973), who provide one of the first studies on efficient wildlife management, which is quite involved by considering both migratory behavior of the waterfowl and the provision of habitats in terms of wetlands for breeding. An unusual approach is used by Cory and Martin (1985), who develop a static model, and construct a production function where elk and cattle compete for land. Stage structured population models often generate pulse harvesting in optimum, implying selective harvesting of certain stage classes in one period followed by no or minor harvest in the next period in order to let the population grow (e.g., Cooper 1993; Naevdal et al. 2012; Elofsson et al. 2017).

Approximately half of the studies consider ecological interaction either in terms of migration (Brown and Hammock 1973; Boman et al. 2003; Skonhoft 2005; Chen and Skonhoft 2013), with livestock (Cory and Martin 1985; Horan and Wolf 2005; Horan et al. 2008) or among game animal (Elofsson et al. 2017). A novel aspect is provided by Naevdal et al. (2012), who consider the animals’ adaptation to hunting through fertility. Hussain and Tschirhart (2010) constitute an unusual study by considering uncertainty in hunting success and interaction between hunters and a wildlife agency.

With respect to included benefits and costs, all studies in Table 3 include hunters’ valuation in terms of recreational, meat, and/or trophy value. A few studies add viewing or existence values (Cooper 1993; Boman et al. 2003). Private costs are considered in all studies, but there is a small difference depending on perspective, from the landowner or hunter. When landowner perspective is used, costs include expenses for supplying populations such as feeding. When hunters are at focus, hunting costs for, e.g., licenses and equipment, are included. It is also shown in Table 3 that several studies consider external costs in terms of browsing, traffic accidents, and transmission of diseases.

Similar to benefit and cost studies, several studies on optimal populations apply their models on game animals, in particular deer, in USA. About half of the studies are applied to countries in Europe, where the optimal population of moose has been at focus. There are very few studies applied on carnivores, and none of those are applied in emerging or developing countries where carnivore hunting can be an important source of income for the residents. All studies are also partial in the sense that they do not simultaneously include ecological interaction, several external costs, and uncertainty, while doing that is indeed a theoretical and empirical challenge.

Wildlife policies

The second main question raised in the literature is on policy design in order to move from actual undesired to desired wildlife population sizes. Wildlife policies are typically motivated by the wildlife species imposing an external cost of some kind, e.g., on agricultural or forestry production, or by the species providing benefits that have public good characteristics, e.g., if the species is threatened and its preservation is considered highly valuable. In principle, externality problems can be solved if all stakeholders cooperate, which has proven successful for natural resource management in relatively small communities (Ostrom 1990). There is a large body of literature on voluntary cooperation on management of natural resources, but we have not found applications to wildlife resources in this regard. However, conditions for successful voluntary cooperation are restrictive, and governmental interventions with coercive power are often needed for improved management.

Table 3 Examples of studies calculating optimal wildlife population sizes

	Animal and study region	Population growth	Ecological interaction ^a	Value items	Cost items
Brown and Hammack (1973)	Waterfowl, Canada	Logistic	Migration to USA, wetlands as habitats	Hunting value ^b	Hunting cost, opportunity cost of habitat
Keith and Lyon (1985)	Deer, USA	Logistic		Hunting value ^b	Hunting cost
Cory and Martin (1985)	Moose, USA	Static model	Interaction with livestock	Hunting value ^b	Hunting permit, lost livestock
Cooper (1993)	Deer, USA	Stage structured model		Hunting value ^b , trophy value, viewing	Hunting permit
Zivin (2000)	Feral pigs, USA	Logistic		Hunting values ^b	Hunting permit, browsing
Boman et al. (2003)	Wolf, Sweden	Logistic	Migration in Sweden	Hunting and existence values	Hunters' cost, predation
Skonhøft (2005)	Moose, Norway	Stage structured model	Migration to Sweden	Landowners' net income from permits	Browsing
Horan and Wolf (2005)	Deer, USA	Logistic	Interaction with livestock	Hunting values ^b	Transmission of disease
Horan et al. (2008)	Deer, USA	Logistic	Interaction with livestock	Hunting values ^b	Transmission of disease, biosecurity measures
Hussain and Tschirhart (2010)	Moose, USA	Logistic		Uncertain hunting values and incomes	Hunting cost
Olausen and Skonhøft (2011)	Moose Norway	Stage structured model		Landowners' net incomes from meat	Browsing, traffic accidents
Naevdal et al. (2012)	Moose, Norway	Stage structured model	Moose adaptation to hunting pressure	Landowners' income from permits ^c	Provision hunting permit
Chen and Skonhøft (2013)	Moose, Norway	Logistic	Migration to Sweden	Landowners income from permit ³	Browsing
Elofsson et al. (2017)	Deer, Sweden	Stage structured model	Fallow and roe deer	Landowners' hunting values	Feeding cost

^a Interaction with habitats and other populations^b Hunting values generally include meat value, recreational, trophy, and other values from hunting^c Price and income of licenses depend on market structure for hunting licenses

Compared with the literature on population size, that on policy design is smaller. In total, we found 32 studies, and some of them (Skonhøft 2005; Olausson and Skonhøft 2011; Chen and Skonhøft 2013) overlap with the literature on optimal population sizes where they compare outcomes of different policy alternatives to reach the optimal population size. In the literature, three types of governmental policies for wildlife have been found and analyzed: (i) distribution of property rights (11 studies), (ii) command and control (4 studies), and (iii) wildlife damage compensation and economic incentives for wildlife damage prevention (14 studies). Similar to the literature on optimal population size, this classification of policies can be compared with the literature on fishery economics, which covers policies in terms of property right (e.g., comparisons between open access and private property regimes, creation of reserves), command and control (e.g., seasonal harvest and gear regulations, restrictions on harvests, and entry restrictions), and economic incentives (landing taxes and tradable fishing quotas) (Clark 1990; Conrad 2010). Each type of policy is associated with transaction costs, such as agents' costs for monitoring and verifying compliance with the formal rules, on which there is a small literature (five studies).

Distribution of property rights

When property rights are well defined, the costs and benefits generated by economic activities can be accounted for on markets where goods and services are traded. Bargaining between different agents could solve the problem of externalities if transaction costs are negligible (Coase 1937) (transaction costs are discussed separately below). In the economics of wildlife literature, the focus is on the potential of private property rights to create markets for wildlife, and the magnitude of negative externalities that are created when private property rights are applied and markets exist for hunting rights.

With respect to the potential of private property rights to solve wildlife management problems, two polar cases of wildlife ownership, public and private, are often analyzed and compared in the literature. It is argued that the provision of private property rights to wildlife provides economic incentives for efficient management of wildlife populations, as it permits property owners to market wildlife in various ways (Leopold 1930). Private property rights have proven successful in increasing the population of elephants in Zimbabwe and South Africa, where markets for ivory and hunting exist (Rasker et al. 1992), but when such markets are not allowed, or demand is limited, private property rights are insufficient to preserve valuable species (Rasker et al. 1992; Tisdell 2004). Another possible reason for failure of private property rights is wildlife species migration, which implies that habitat management by one land owner could positively affect wildlife

abundance on neighboring land for which the landowner is not rewarded (Tisdell 2004). Also, there can be external effects because private landowners with hunting rights may ignore the risk for wildlife-vehicle collisions (Olausson and Skonhøft 2011). Further, private property rights to wildlife could be inefficient if there are high costs for monitoring, and hence a higher risk for illegal hunting, albeit we have not found studies on these topics.

Several studies have investigated demand and supply for hunting (Sun et al. 2005; Poudyal et al. 2012), but few have considered how well these markets work in the presence of market failures such as externalities, market power, and imperfect information. When market failure occurs in terms of species' migratory behavior, Skonhøft (2005) and Nilsen et al. (2009) showed that private property regimes imply lower profits from moose hunting compared to the socially optimal solution since each landowner considers the game on their land and not the impact on other landowners. Nilsen et al. (2009) further showed that the difference in total profits between the two regimes can be either reinforced or weakened by the consideration of additional externalities in terms of browsing damage and wolf predation, depending on their spatial allocation. The implications of landowners' market power on the market for moose hunting licenses were examined by Chen and Skonhøft (2013). They showed that in the presence of externalities in terms of wildlife migration and forest damage, market power can create substantial economic losses for society, and also reduces the size of the wildlife population. Table 4 summarizes the different property right configurations, and the causes and consequences of success and failure associated with different allocations of property rights.

Command and control policies

The main type of command and control policy investigated in the literature and implemented in practice is restrictions on hunting imposed by the government or an authorized agency, which implies a limitation of the use of property rights. Hunting regulations applied include restrictions on the quantity of game harvest and/or restrictions on how, where, and when to hunt. Further, there are regulations on the distribution of monetary revenues from wildlife. Despite the common use of regulations, we found only four studies addressing their performance.

Restrictions on harvesting are commonly assumed when threatened species are protected by law, thereby giving them a chance to re-establish. One example is Skonhøft (2006), who investigated the positive and negative economic consequences of the legal protection of wolf in Scandinavia under four rules for the stock and harvest for moose (constant stock, proportional harvest, fixed harvest, and optimal harvest and stock), which is the wolf's main prey, given a secondary impact on moose-vehicle collisions and forest browsing. The

Table 4 Different property rights configurations and associated effects investigated in the reviewed literature

	Chen and Skonhøft (2013)	Gibson and Marks (1995)	Johannessen and Skonhøft (2004)	Johannessen and Skonhøft (2009)	Nilsen et al. (2009)	Olaussen and Skonhøft (2011)	Poudyal et al. (2012)	Rasker et al. (1992)	Skonhøft (2005)	Sun et al. (2005)	Tisdell (2004)
Hunting property right	X	X	X	X	X	X	X	X	X	X	X
Land owner has rights			X								
Park has rights			X					X			
Government has rights ^a		X						X			
Benefit sharing		X	X								
Socially optimal solution					X	X			X		
Browsing/grazing damage	X	X		X	X	X		X	X		
suffered by victim											
Voluntary agreements on damage management is possible				X				X			
Habitat improvements											X
Migration	X				X		X	X	X		
Vehicle collisions						X					

Articles that only deal with command-and-control regulation and damage compensation/performance payments are not included

^a The alternative that a public body has the right to decide on harvesting is considered, but need not imply that the socially optimal harvesting strategy is chosen

theoretical results do not lead to any clear conclusions with respect to the ranking of these rules, but the empirical application to moose management in Norway showed that the overall net benefits of the different harvesting regimes depend strongly on the cost of vehicle collisions. Seasonal restrictions on hunting are investigated in Skonhøft and Olaussen (2005), with an application to moose, which migrate and create hunting revenues but also cause damage for forest owners at a magnitude that differs between regions. The results show that an extension of the hunting season to include the winter season increases total social net gains, because it allows for more hunting in the region with the most damage to forest.

Regulations can also apply to the distribution of the revenues from wildlife. Gibson and Marks (1995) evaluated a system with benefit sharing between local communities for the case of wildlife in Africa, and concluded that this has not been effective in preventing poaching. Instead, observed reductions in poaching can merely be attributed to the increased efforts to enforce laws prohibiting poaching. On the other hand, Johannessen and Skonhøft (2004) showed that an integrated conservation and development projects (ICDPs), which are frequently set up in African countries, can improve wildlife conservation. They applied game theory with two players: a park manager who benefits from tourism and hunting and a group of local people who benefit from illegal hunting, but also bear the cost of wildlife damage.

In summary, there are few economic studies on command and control regulations, mainly theoretical, and even fewer empirical studies. We have not found studies on regulations of how to hunt (e.g., gear restrictions and restrictions on the organization of hunting).

Wildlife payment programs and economic incentives for wildlife damage prevention

The analysis of economic instruments has received more attention in the literature than that of command and control regulations. Instead of restrictions on hunting, economic instruments make it more costly or beneficial for hunters to increase or decrease harvest depending on desired direction of change in population size. In total, 14 studies are included in this review and the economic instruments applied are damage compensation, subsidies to damage prevention, subsidies to culling of damaging species, and voluntary payments for wildlife-enhancing measures.

Payments can be made for damage caused by wildlife (compensation payments) or for the presence of wildlife within a certain area (performance payments). Compensation programs are most common and are applied in practice all over the world. They involve pastoralists and farmers affected by wildlife damage being given the right to compensation for killed and injured animals and damage to crops (Bulte and Rondeau 2005). Compensation programs are relatively cheap

to implement in poverty-stricken areas and therefore popular among conservationists and governments (Bulte and Rondeau 2007). The decision to introduce a compensation scheme can be motivated by wildlife damage threatening the livelihood of farmers, while at the same time, abatement methods are considered too costly or unethical, or threaten the existence of protected species (Yoder 2000).

Many studies point out that compensation programs not only constitute a transfer of ownership rights and, hence, wealth in the sense of Coase (1937) but also create economic incentives which affect the behavior of agents (Yoder 2000; Bulte and Rondeau 2005). Maclellan et al. (2009) argue that a positive consequence of a compensation program in southern Kenya is that it has helped reduce the number of retaliatory killings of lions. On the negative side, several studies note that compensation schemes reduce private incentives to protect livestock (Bulte and Rondeau 2005; Zabel et al. 2011). Rollins et al. (1996) therefore suggest that predation payments should be conditional on observed abatement efforts by farmers. Moreover, if compensation levels are set at a high level, compensation can have an effect similar to subsidies for farming and livestock, providing incentives for entry into the sector and thus increasing the number of producers operating in the area (Rollins et al. 1996; Nyhus et al. 2005; Bulte and Rondeau 2005). This can increase wildlife damage if it increases the amount of prey available to predators. Furthermore, if livestock is free-ranging, an increase in livestock can lead to stronger competition between livestock and wildlife for food and space, which can reduce stocks of wildlife (Bulte and Rondeau 2005). Compensation schemes can also involve increased incentives for reallocating labor from defensive hunting to farming, which could lead to further decreases in wildlife habitats (Rondeau and Bulte 2007).

Rollins et al. (1996) note that a public compensation scheme could, at least hypothetically, function as an insurance scheme, where the government faces a problem similar to that of an insurance company, i.e., to provide compensation contracts while assuring that farmers still undertake relevant damage prevention measures, such that total compensation costs can be kept as low as possible. Mishra et al. (2003) report on a program intended to provide incentives to manage snow leopard predation. Farmers were asked to give up a share of grazing land to ensure the availability of natural prey for the snow leopard and were given compensation in return for the land abandoned. This compensation was then used to create a communal insurance fund to offset the costs of livestock losses.

Performance payment is an alternative to compensation for killed livestock or crop damage where payments are conditioned on the abundance of wildlife on the landowner's property (Ferraro and Kiss 2002). By conditioning payments on wildlife abundance, performance payment schemes have a more clear aim to provide incentives for wildlife management compared to compensation schemes, while a reallocation of

property rights occurs under both schemes. The only performance payment program currently in operation is that in Sweden for lynx and wolverine populations maintained on reindeer herders' land (Zabel et al. 2011). Performance payments are paid based on the expected offspring, and are intended to cover the expected costs of damage. Zabel et al. (2011) show that the relative cost of ordinary compensation and performance payment programs depends on the relationship between predator and prey. The policy requires identification of eligible groups and a mechanism for distributing the compensation among individuals in affected groups (Zabel and Engel 2010; Zabel et al. 2011). Similar to compensation schemes, performance payments create incentives for reduced poaching and increased entry into the affected sector. However, in contrast to compensation schemes, they do not reduce incentives for preventive efforts which reduce the wildlife damage (Zabel et al. 2011).

Economic instruments are also used to provide incentives for abatement of wildlife damage. Abatement methods that reduce the damage caused by wildlife can be graded according to the harm caused to the animals, where hunting is the most harmful abatement method (Rollins et al. 1996; Yoder 2000). Public policies that control abatement include incentives for changes in hunting pressure, incentives for non-lethal abatement methods such as translocation of problem animals, scare devices, guard dogs, barriers, and improved livestock husbandry (Sandry et al. 1983; Rollins et al. 1996; Breck and Meier 2004). Abatement strategies for reducing damage have been examined by means of cost-benefit analysis, e.g., for supplemental feeding programs for black bears to reduce forestry damage (Ziegler 2006), and cost-efficiency analysis, e.g., with regard to the trade-off between fox culling and various prevention measures for sheep farms (Moberly et al. 2004). In the latter study, it is shown that the optimal solution varies between farm types and locations, implying that a uniform policy can be expensive compared with a differentiated one. As observed by Rollins et al. (1996), prevention of wildlife damage can give rise to spatial externalities; an increase in prevention efforts by an individual farmer can mean that neighboring farmers experience an increase in predation and at worst there is no net gain to society.

Berger (2006) evaluated the impact of predator culling subsidies 1939–1980 on the profits of the sheep industry. The results show that the impact of the subsidies is small compared with that of output prices and production costs, and the author therefore argues that damage compensation is more efficient than subsidies to predator control in ensuring the profitability of the sheep industry.

There are substantial voluntary payments to individuals and groups for supplying wildlife or wildlife habitats (Ferraro and Simpson 2002). Such schemes operate both within countries and across country borders. Many of these payments are channeled through eco-friendly “products,” such as

eco-tourism, but Ferraro and Simpson (2002) argue that paying directly for ecosystem habitat protection, such as rainforest preservation, would be more cost-effective.

In summary, much of the literature on damage compensation and prevention is mainly theoretical, while there are few empirical studies that shed light on the magnitude of different positive and negative effects that are foreseen by the theoretical work. In particular, this applies to the effects of subsidies to damage prevention, where the amount of spending can be considerable.

Transaction cost

If the social planner had complete and perfect information, any of the presented policies could be designed to give efficient solutions, which is not the case in practice. Uncertainties are associated with the measurement of population sizes, damages, and benefits. Some of these uncertainties are shared by all actors, such as uncertainty regarding the impact of climate change on wildlife population sizes. Other uncertainties are asymmetrically distributed, where, e.g., the size of damage from predation is likely to be known by the affected farmer but uncertain to a regulator paying the compensation scheme. These uncertainties give rise to transaction costs for collection of information, meetings prior to making decision on regulations, and costs of monitoring and supervision of compliance of implemented policies.

There is a relatively large body of literature on transaction costs of payments for ecosystem services, in particular for biodiversity management (see review in Vatn 2010), but the applications to wildlife management are scarce (Mburu and Birner 2002; Mburu et al. 2003; Schwerdtner and Gruber 2007). Mburu and Birner (2002) show that the benefit-cost ratio for landowners, including transaction cost, is highly dependent on who participates in the co-management of wildlife in Kenya. Mburu et al. (2003) show that these transaction costs correspond to 13% of the total provision costs of wildlife, and are determined by both socioeconomic and ecological conditions. Schwerdtner and Gruber (2007) compare ex ante and ex post compensation schemes for damages by the European otter in Germany. They show that transaction costs of both systems increase with high dispersal of the damages in space and time, but are higher under an ex post scheme since compensation is determined on a case-by-case basis and incentives for overstating the damages are higher, while the ex ante compensation is based on prediction and modeling of damages. The calculated transaction costs of an ex ante and ex post compensation scheme for damages of otter on fish farms in Germany correspond to 22 and 35%, respectively, of the total costs.

Transaction costs and uncertainty can be reduced by appropriate design of policies. Abildtrup and Jensen (2014) suggest a combined efficient tax/subsidy scheme on an

observable control variable, the game population, and non-observable variable, harvest, from the regulator's point of view. Hunters have to pay a tax per animal when the population is above the targeted population, and obtain payment when the population is below the target. This analysis is extended by Jensen et al. (2016), who assume that neither population size nor hunting bags are known by the regulator, who instead calculates and regulates population size based on hunters' bag reports. Taxes are introduced on both bag reports and estimated population size. A low report on bags reduces tax payments, but results in higher calculated population size (because of the reported low hunting pressure). The hunter thus faces a trade-off between paying taxes for reported bags and the risk of paying a tax on the populations.

Further understanding of the magnitude of uncertainty and transaction costs associated with different policy choice is likely to be valuable, as well as studies that relate these effects to the distributional outcome of policies. Considering the prevalence of different kinds of uncertainty in wildlife management, analyses of clever policy design to mitigate costs of uncertainty and transaction costs are in particular need.

Discussion and conclusions

This brief overview showed that three fourths of all included studies calculate costs and benefits of different game animals. However, in order to determine whether a population should increase or decrease, the benefit and cost estimates must be related to the number of game animals, which was lacking in almost all studies. Furthermore, almost all studies calculated only one type of benefit or cost, usually in terms of the average value or cost for one animal. The included benefits or costs were expressed in different ways which makes it difficult to compare results between studies. Certain types of costs were seldom estimated, such as costs for farmers other than the direct cost due to the loss of livestock or crop, and costs that arise due to dispersal effects in the economy. Benefits associated with wildlife's contribution to biodiversity and associated effects on provision of ecosystem services were not considered. We therefore agree with Core and Martin (1985) that "The value of a day or a trip must be converted to the value of the marginal animal. Otherwise, the (recreational, our parenthesis) value estimate—no matter how precise empirically or theoretically—has little management value" (pp. 283). Similar reasoning can be applied for estimates of wildlife costs.

Nevertheless, benefit and cost estimates can be of importance for decisions on specific projects, such as creation of parks for protecting wildlife or compensation payments for wildlife damage. There is an agreement in the scientific community on choice of appropriate methods for estimating benefits and costs, but the estimated numbers differ considerably

across different studies. With respect to benefit estimates, value from hunting was the most commonly estimated benefit type and the values reported varied between 41 and 545 USD/hunting day, measured in 2013 prices. The most frequently studied animals are deer and moose, which accounted for 65% of all studies. A majority of the valuation studies, 70%, are applied to USA and the rest to Europe and Australia. The studies estimating costs of wildlife focused on damages from livestock predation by carnivores and crop destruction by ungulates. A few studies calculate the costs of traffic collisions and transmission of diseases to animals and humans. Costs were measured mainly on a per household basis, and showed a wide range, from 2 to 9200 USD/farm household. It is interesting to note the relatively large number of cost studies in developing countries and emerging economies, 35%, compared with the lack of studies on wildlife benefits in these countries. Explanation can be the costs of livestock predation borne by farmers close to nationally protected areas in several developing countries and the lack of private property rights to wildlife which implies smaller possibilities to exploit wildlife benefits.

With respect to the second main question, the literature on policy design for wildlife management identifies three main mechanisms for solving conflicts arising from asymmetric distribution of costs and benefits: redistribution of property rights, command and control policies for protecting endangered wildlife species, and economic instruments. The economic instruments include compensation or performance payments for wildlife populations or abatement of damage, and economic incentives for wildlife damage prevention. Most studies, 25 of a total of 32 studies on policy design examine effects of redistribution of property rights or economic instruments. A few studies calculate transaction costs or address policy design under conditions of uncertainty.

There is a common agreement that redistribution of property rights and command and control instruments will not function in isolation if the market value of wildlife is low or species migrate. Also, these mechanisms do not significantly reduce incentives for those who experience the costs of wildlife to decrease the population by illegal hunting and killing. Compensation payments, funded by taxes or hunting fees, are therefore needed. These payments can be based on stated losses, market values of livestock/crop, or performance of wildlife. Compensation payments based on stated damage costs might be too high, since they create an incentive to overstate the actual costs of the loss. An alternative is to base the payment on the market value of the livestock/crop loss, but this does not account for other costs affecting farmers due to, e.g., productivity losses, increased labor, and measures for damage prevention. A feature common to both these forms of payment is that they create disincentives to invest in preventive measures. Performance payments are preferable, but are difficult to implement and enforce, which might result in high transaction costs

for designing and monitoring. The few studies on transaction costs showed that these costs can be considerable and depend on type of compensation scheme.

Overall, a conclusion from this overview is that a relatively large body of literature has developed since the 1970s in particular on the calculation of costs and benefits of game species. The concentration of applications to a few species mainly in the USA and Europe calls for future studies on different game animals in other parts of the world. Moreover, there is a lack of studies comparing costs and benefits in a wider setting including a broader set of cost and benefit items. One can also note the richness with respect to theoretical findings in the relatively small literature on policy design. However, to understand the importance of these findings for practical policy, empirical applications are necessary, and require information on hunters' responses to different regulation schemes. This, in turn, necessitates data on the relation between animal populations, harvests, values, and costs. Such studies remain to be done: population estimates are in general uncertain, and in general, the literature on costs and benefits of wildlife does not relate these items to number of animals. Policy design would then need to account for uncertainty in estimates of population sizes, values, and costs, which was made only by a few theoretical studies. One can further note that there seems to be almost no analysis of the prospects for, or existence of, alternatives to wildlife damage compensation schemes, such as private insurance schemes or changes in liability rules. Finally, studies are also needed to evaluate the functioning of the many existing policies in order to improve and fine-tune their efficiency.

Acknowledgements Valuable comments from three anonymous reviewers of this journal are gratefully acknowledged. The work was funded by the Swedish Environmental Protection Agency under the grant numbers 11/96 and 12/133.

Open Access This article is distributed under the terms of the Creative Commons Attribution 4.0 International License (<http://creativecommons.org/licenses/by/4.0/>), which permits unrestricted use, distribution, and reproduction in any medium, provided you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons license, and indicate if changes were made.

References

- Abay GY, Bauer H, Gebrihiwot K, Deckers J (2011) Peri-urban spotted hyena (*Crocuta crocuta*) in northern Ethiopia: diet, economic impact, and abundance. *Eur J Wildl Res* 57:759–765
- Abildtrup J, Jensen F (2014) The regulation of hunting. A game population based tax on hunters. *Rev Agr Environ Stud* 95:281–298
- Baker PJ, Boitani L, Harris S, Saunders G, White PCL (2008) Terrestrial carnivores and human food production: impact and management. *Mammal Rev* 38:123–166

- Barnes JJ, Schier C, Van Rooy G (1999) Tourists' willingness to pay for wildlife viewing and wildlife conservation in Namibia. *S Afr J Wildl Res* 29:101–111
- Berger KM (2006) Carnivore-livestock conflicts: effects of subsidized predator control and economic correlates on the sheep industry. *Conserv Biol* 20:751–761
- Boman M, Bostedt G, Persson J (2003) The bioeconomics of the spatial distribution of an endangered species: the case of the Swedish wolf population. *J Bioecon* 5:55–74
- Boyle KJ, Bishop RC (1987) Valuing wildlife in benefit-cost analyses: a case study involving endangered species. *Water Resour Res* 23:943–950
- Breck S, Meier T (2004) Managing wolf depredation in the United States: past, present, and future. USDA National Wildlife Research Center, Staff Publications, Paper 83. Available at http://digitalcommons.unl.edu/cgi/viewcontent.cgi?article=1081&context=icwdm_usdanwrc. Accessed 13 Sept 2016
- Brown GM, Hammack J (1973) *Rev Econ Stat* 55:73–82
- Bullock CH, Elston DA, Chalmers NA (1998) An application of economic choice experiments to a traditional land use—deer hunting and landscape change in the Scottish highlands. *J Environ Manag* 52:335–351
- Bulte EH, Rondeau D (2005) Why compensating wildlife damages may be bad for conservation. *J Wildl Manag* 69:14–19
- Bulte E, Rondeau D (2007) Compensation for wildlife damages: habitat conversion, species preservation and local welfare. *J Environ Econ Manag* 54:311–322
- Burger LW, Miller DA, Southwick RI (1999) Economic impact of northern bobwhite hunting in the southeastern United States. *Wildl Soc Bull* 27:1010–1018
- Butler JRA (2000) The economic costs of wildlife predation on livestock in Gokwe communal land, Zimbabwe. *Afr J Ecol* 38:23–30
- Chen W, Skonhofs A (2013) On the management of interconnected wildlife populations. *Nat Resour Model* 26:1–25
- Ciriacy-Wantrup SV (1947) Capital returns from soil-conservation practices. *J Farm Econ* 29:1181–1196
- Clark CW (1990) Mathematical bioeconomics: the optimal management of renewable resources, 2nd edn. Wiley, New York
- Coase RH (1937) The nature of the firm. *Economica* 4:386–405
- Conover MR, Pitt WC, Kessler KK, Dubow TJ, Sandborn WJ (1995) Review of human injuries, illnesses, and economic losses caused by wildlife in the United States. *Wildl Soc Bull* 23:407–414
- Conrad JM (2010) Resource economics. Cambridge University Press, Cambridge
- Cooper J (1993) A bioeconomic model for estimating the optimal level of deer and tag sales. *Environ Resour Econ* 3:563–579
- Cory DC, Martin WE (1985) Valuing wildlife for efficient multiple use: elk versus cattle. *West J Agric Econ* 10:282–293
- Delibes-Mateos M, Giergiczny M, Caro J, Viñuela J, Riera P, Arroyo B (2014) Does hunters' willingness to pay match the best hunting options for biodiversity conservation? A choice experiment application for small-game hunting in Spain. *Biol Conserv* 177:36–42
- Desvougues WH, Smith VK, McGivney MP (1983) A comparison of alternative approaches for estimating recreation and related benefits of water quality improvements. Report for the U.S. Environmental Protection Agency, Economic Analysis Division, Washington D.C.
- Elofsson K, Mensah JT, Kjellander P (2017) Optimal management of two ecologically interacting deer species—reality matters, beliefs don't. *Nat Resour Model* 30:1–24
- Ferraro P, Kiss A (2002) Direct payments to conserve biodiversity. *Science* 298:1718–1719
- Ferraro P, Simpson R (2002) The cost-effectiveness of conservation payments. *Land Econ* 78:339–353
- Fried B, Adams R, Berrens R, Bergland O (1995) Willingness to pay for a change in elk hunting quality. *Wildl Soc Bull* 23:680–686
- Gibson CC, Marks SA (1995) Transforming rural hunters into conservationists: an assessment of community-based wildlife management programs in Africa. *World Dev* 23:941–957
- Gordon IJ, Hester AJ, Festa-Bianchet M (2004) Review: the management of wild large herbivores to meet economic, conservation and environmental objectives. *J Appl Ecol* 41:1021–1031
- Grado S, Hunt K, Whiteside M (2007). The economic impact of white-tailed deer hunting in Mississippi. *Proc Annu Conf Southeastern Assoc Fish Wildl Agencies* 61: 58–67
- Gren IM (2008) Economics of alien species: choosing targets and policies. *Boreal Environ Res* 13:17–32
- Gren IM, Jägerbrand A (2017). Costs of animal-vehicle collisions with ungulates in Sweden. Working paper 2017:3, Department of Economics, Swedish University of Agricultural Sciences, Uppsala, Sweden. Available at <https://pub.epsilon.slu.se/14592/>. Accessed 17 Oct 2017
- Gusset M, Swarner M, Mponwane L, Keletile K, McNutt J (2009) Human-wildlife conflict in northern Botswana: livestock predation by endangered African wild dog *Lycaon pictus* and other carnivores. *Oryx* 43:67–72
- Häggmark-Svensson T, Engelman M, Elofsson K, Gren I-M (2015) Values, costs and policies for wildlife management: A review. Working Paper 2015:01, Department of Economics, Swedish University of Agricultural Sciences, Uppsala. At https://ideas.repec.org/p/hhs/slueko/2015_001.html. Accessed 17 Aug 2016
- Hofer D (2002) The lion's share of the hunt. Trophy hunting and conservation—a review of the legal Eurasian tourists hunting market and trophy trade under CITES. Brussels: TRAFFIC Europe Regional Report. Available at www.traffic.org/general-reports/traffic_pub_gen9.pdf. Accessed 18 Aug 2016
- Horan RD, Wolf CA (2005) The economics of managing infectious wildlife disease. *Am J Agric Econ* 87:537–551
- Horan RD, Wolf CA, Fenichel EP, Mathews KH (2008) Joint management of wildlife and livestock disease. *Environ Resour Econ* 41:47–70
- Horne P, Petäjästö L (2003) Preferences for alternative moose management regimes among Finnish landowners: a choice experiment approach. *Land Econ* 79:472–482
- Howery L, DeLiberto T (2004) Indirect effects of carnivores on livestock foraging behavior and production. *Sheep Goat Res J* 19:53–57
- Huijser MP, McGowen P, Fuller J, Hardy A, Kociolek A, Clevenger P, Smith D, Ament R, (2008) Wildlife-vehicle collision reduction study: Report to congress. Western Transportation Institute, Montana State University. Available at <https://www.fhwa.dot.gov/publications/research/safety/08034/08034.pdf>. Accessed 27 Jul 2017
- Hussain A, Tschirhart JJ (2010) Optimal harvest licensing when harvest success is uncertain. *Am J Agric Econ* 92:125–140
- Ikeda N (2004) Economic impacts of livestock depredation by snow leopard *Uncia uncia* in the Kanchenjunga conservation area, Nepal Himalaya. *Environ Conserv* 31:322–330
- Jensen F, Jacobsen JB, Jellesmark Thorsen B (2016) Designing hunting regulation under population uncertainty and self-reporting. *J For Econ* 24:157–171
- Johannessen AB, Skonhofs A (2004) Property rights and natural resource conservation. A bio-economic model with numerical illustrations from the Serengeti-Mare ecosystem. *Environ Resour Econ* 28:469–488
- Johannessen AB, Skonhofs A (2009) Local common property exploitation with rewards. *Land Econ* 8:637–654
- Jones K (2004) Economic impact of sheep predation in the United States. *Sheep Goat Res J* 19:5–12
- Keith J, Lyon K (1985) Valuing wildlife management: a Utah deer herd. *West J Agric Econ* 10:216–222

- Laporte I, Muhly T, Pitt J, Alexander M, Muisani M (2010) Effects of wolves on elk and cattle behaviors: implications for livestock production and wolf conservation. *PLoS One* 5:8
- Leopold A (1930) Report to the American game conference on an American game policy. *Trans Conf Am Game* 17:284–309
- Lindsey P, Alexander R, Frank L, Mathieson A, Románach S (2006) Potential of trophy hunting to create incentives for wildlife conservation in Africa where alternative wildlife-based land uses may not be viable. *Anim Conserv* 9:283–291
- Livengood KR (1983) Value of big game from markets for hunting leases: the hedonic approach. *Land Econ* 59:287–291
- Loomis JB, White DS (1996) Economic benefits of rare and endangered species: summary and meta-analysis. *Ecol Econ* 18:197–206
- Loomis J, Cooper J, Alien S (1988) The Montana elk hunting experience: a contingent valuation assessment of economic benefits to hunters. (Helena, MT): Montana Department of Fish, Wildlife and Parks. Available at <https://archive.org/details/montanaelkhuntin00loomrich>. Accessed 14 Aug 2016
- Loomis J, Donnelly D, Sorg-Swanson C (1989) Comparing the economic value of forage on public lands for wildlife and livestock. *J Range Manag* 42:134–138
- MacLennan SD, Groom RJ, Macdonald DW, Frank LG (2009) Evaluation of a compensation scheme to bring about pastoralist tolerance of lions. *Biol Conserv* 142:2419–2427
- Madhusudan MD (2003) Living amidst large wildlife: livestock and crop depredation by large mammals in the interior villages of Bhadra tiger reserve, South India. *Environ Manag* 31:0466–0475
- Mäler K-G (1974) Environmental economics: a theoretical inquiry. Johns Hopkins University for Resources for the Future, Baltimore
- Marbuah G, Gren I-M, McKie B (2014) Economics of harmful invasive species: a review. *Diversity* 6:500–523
- Mawdsley J, O'Malley R, Ojima D (2009) A review of climate-change adaptation strategies for wildlife management and biodiversity conservation. *Conserv Biol* 29:1080–1089
- Mburu J, Bimer R (2002) Analyzing the efficiency of collaborative wildlife management: the case of two community wildlife sanctuaries in Kenya. *J Org Theory Behav* 5:359–394
- Mburu J, Bimer R, Zeller M (2003) Relative importance and determinants of landowners' transaction costs in collaborative wildlife management in Kenya: an empirical analysis. *Ecol Econ* 45:59–73
- Mensah J, Elofsson K (2017) An empirical analysis of hunting lease pricing and value of game in Sweden. *Land Econ* 93:292–308
- Michalski F, Boulhosa RLP, Faria A, Peres CA (2006) Human–wildlife conflicts in a fragmented Amazonian forest landscape: determinants of large felid depredation on livestock. *Anim Conserv* 9:179–188
- Mishan EJ (1971) Evaluation of life and limb: a theoretical approach. *J Polit Econ* 79:687–705
- Mishra C (1997) Livestock depredation by large carnivores in the Indian trans-Himalaya: conflict perceptions and conservation prospects. *Environ Conserv* 24:338–343
- Mishra C, Allen P, McCarthy T, Madhusudan MD, Bayarjargal A, Prins HHT (2003) The role of incentive programs in conserving the snow leopard. *Conserv Biol* 17:1512–1520
- Mitchell R C, Carson R T (1989) Using surveys to value public goods: the contingent valuation method. Resources for the Future, Washington DC.
- Moberly RL, White PCL, Webbon CC, Baker PJ, Harris S (2004) Modelling the costs of fox predation and preventive measures on sheep farms in Britain. *J Environ Manag* 70:129–143
- Morton KM, Adamowicz WL, Boxall PC (1995) Economic effects of environmental quality change on recreational hunting in northwestern Saskatchewan: a contingent behavior analysis. *Can J For Res* 25: 912–920
- Naevdal E, Olausson JO, Skonhoft A (2012) A bioeconomic model of trophy hunting. *Ecol Econ* 73:194–205
- Nilsen EB, Skonhoft A, Mysterud A, Milner J, Solberg EJ, Andreassen HP, Stenseth NC (2009) The role of ecological and economic factors in the management of a spatially structured moose *Alces alces* population. *Wildl Biol* 15:10–23
- Nyhus P, Osofsky S, Ferraro PJ, Madden F, Fischer H (2005) Bearing the costs of human–wildlife conflict: the challenges of compensation schemes. In: Woodroffe R (ed) *People and wildlife, conflict or co-existence?* Cambridge University Press, Cambridge, pp 107–121
- Olausson JO, Skonhoft A (2011) A cost-benefit analysis of moose harvesting in Scandinavia. A stage structured modelling approach. *Resour Energy Econ* 33:589–611
- Ostrom E (1990) *Governing the commons*. Cambridge University Press, New York
- Pack S, Golden R, Walker A (2013) Comparison of national wildlife management strategies: what works where, and why? Heinz Center for Science, Economics & Environment. Available at https://www.academia.edu/4059587/Comparison_of_national_wildlife_management_strategies_what_works_where_and_why. Accessed 19 Aug 2016
- Poudyal NC, Bowker JM, Green GT, Tarrant MA (2012) Supply of private acreage for recreational deer hunting in Georgia. *Hum Dimens Wildl* 17:141–154
- Ramler J, Hebblewhite M, Kellenberg D, Sime C (2014) Crying wolf? A spatial analysis of wolf location and depredation on calf weight. *Am J Agric Econ* 96:631–656
- Rao KS, Maikhuri RK, Nautiyal S, Saxena KG (2002) Crop damage and livestock depredation by wildlife: a case study from Nanda Devi biosphere reserve, India. *J Environ Manag* 66:317–327
- Rashford S, Foulke T, Taylor D (2010) Ranch-level economic impacts of predation in a range livestock system. *Rangelands* 32:21–26
- Rasker R, Martin M, Johnson R (1992) Economics: theory versus practice in wildlife management. *Conserv Biol* 6:338–349
- Reed W (1980) Optimal age-specific harvesting in a nonlinear population model. *Biometrics* 36:579–593
- Roemer DM, Forrest SC (1996) Prairie dog poisoning in northern Great Plains: an analysis of programs and policies. *Environ Manag* 20: 349–359
- Rollins K, Briggs I, Hugh C (1996) Moral hazard, externalities, and compensation for crop damages from wildlife. *J Environ Econ Manag* 31:368–386
- Rondeau D, Bulte E (2007) Wildlife damage and agriculture: a dynamic analysis of compensation schemes. *Am J Agric Econ* 89:490–507
- Sandry RA, Buccola ST, Brown GW (1983) Pricing policies for antlerless elk hunting permits. *Land Econ* 59:432–443
- Schwabe K, Schuhmann P, Boyd R, Doroodian K (2001) The value of changes in deer season length: an application of the nested multinomial logit model. *Environ Resour Econ* 19:131–147
- Schwerdtner K, Gruber B (2007) A conceptual framework for damage compensation schemes. *Biol Conserv* 134:354–360
- Shaw W W (1984). Problems in wildlife valuation in natural resource management. In: Petersen GL and Randall A (eds) 'Valuation of wildland resource benefits'. Westview Press, Boulder & London, pp. 221–229
- Shwiff SA, Anderson A, Cullen R, White PCL, Shwiff SS (2013) Assignment of measurable costs and benefit to wildlife conservation projects. *Wildl Res* 40:134–141
- Skonhoft A (2005) The costs and benefits of a migratory species under different management schemes. *J Environ Manag* 76:167–175
- Skonhoft A (2006) The costs and benefits of animal predation: an analysis of Scandinavian wolf re-colonization. *Ecol Econ* 58:830–841
- Skonhoft A, Olausson J-O (2005) Managing a migratory species that is both a value and a pest. *Land Econ* 81:34–50
- Steele J, Rashford B, Foulke T, Tanaka J, Taylor D (2013) Wolf (*Canis Lupus*) predation impacts on livestock production: direct effects, indirect effects, and implications for compensation ratios. *Rangel Ecol Manag* 66:539–544

- Sun L, Van Kooten GC, Voss GM (2005) Demand for wildlife hunting in British Columbia. *Can J Agric Econ* 53:25–46
- Tamang B, Baral N (2008) Livestock depredation by large cats in Bardia National Park, Nepal: implications for improving park–people relations. *Int J Biodivers Sci Ecosyst Serv Manag* 4:44–53
- Taylor JP, Workman P, Bowns J (1979) The economics of sheep predation in western Utah. *J Range Manag* 32:317–321
- Tietenberg T (1997) *Environmental economics and policy*, 2nd edn. Addison-Wesley, Reading
- Tisdell CA (2004) Economic incentives to conserve wildlife on private lands: analysis and policy. *Environmentalist* 24:153–163
- Treves A, Karanth KU (2003) Human-carnivore conflict and perspectives on carnivore management worldwide. *Conserv Biol* 17:1491–1499
- Vatn A (2010) An institutional analysis of payments for ecosystem services. *Ecol Econ* 69:1245–1252
- Woodroffe R, Ginsberg JR (1998) Edge effects and the extinction of populations inside protected areas. *Science* 280:2126–2128
- Yarrow G (2009) Wildlife and wildlife management. *For Nat Resour* 36: 1–4
- Yoder JK (2000) Damage abatement and compensation programs as incentives for wildlife management on private land. In: Clark L, Hone J, Shivik JA, Watkins RA, Vercauteren KC, Yoder JK (eds) *Human conflicts with wildlife: economic considerations*. Proceedings of the third NWRC special symposium. National Wildlife Research Center, Fort Collins, pp 17–28
- Zabel A, Engel S (2010) Performance payments: a new strategy to conserve large carnivores in the tropics? *Ecol Econ* 70:405–412
- Zabel A, Pittel K, Bostedt G, Engel S (2011) Comparing conventional and new policy approaches for carnivore conservation—theoretical results and application to tiger conservation. *Environ Resour Econ* 48:287–301
- Ziegltrum GJ (2006) Cost-effectiveness of the black bear supplemental feeding program in western Washington. *Wildl Soc Bull* 34:375–379
- Zivin J (2000) Managing a multiple-use resource: the case of feral pig management in California rangeland. *J Environ Econ Manag* 39: 189–204