Valuation of large carnivores and regulated carnivore hunting

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Abstract

Large carnivores are keystone species but represent economic costs to hunters. In Sweden, carnivore territories generally overlap with hunting areas, and as a result, conflicts occur because of the competition for prey. The wolf, lynx, and brown bear are protected species by law but are hunted when authorities allocate license hunting quotas. The aim of these quotas is to limit carnivore numbers. We estimate a hedonic model using ordinary least squares to address the effect of large carnivore occurrence on hunting lease prices by accounting for the presence of license hunting quotas for predators. This result is compared with the least absolute deviation estimations, which reduce the influence of outliers in the survey data. To isolate the effect of carnivores on hunting lease prices, we use snow depth and forest productivity as proxy variables for game harvest in the absence of carnivores. Our results show that lynx and wolf presence reduce hunting lease prices, but lynx presence shows higher significance and robustness. Based on median regressions, the marginal implicit price of an additional wolf territory is about 15% larger than that of an additional lynx territory. In contrast, we found no conclusive evidence that bear abundance directly affects hunting lease prices, but regulated bear hunting is found to have a positive and significant impact on hunting leases, suggesting indirect positive net benefits of increased brown bear abundance.

Keywords: Large carnivores · Licensed carnivore-hunting · Least absolute deviations · Hunting lease prices · Hedonic pricing

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1. Introduction and motivation

In most European countries, hunting is a fundamental right that belongs to the land owner. Hunting areas, in particular on forest-dominated land, can provide excellent habitat for ungulate game and large predators preying on those. The wolf (*Canis lupus*), lynx (*Lynx lynx*), and brown bear (*Ursus arctos*) are considered keystone species with a potentially important ecological influence (Paine 1995). However, they also compete for game with human hunters (Heberlein & Ericsson 2008), kill domestic animals, cause fear, and occasionally attack humans (Bergman & Åkerberg 2006, Olson et al. 2014, Frank et al. 2015). Therefore, large carnivores can reduce hunters' willingness to pay (WTP) for renting hunting grounds where carnivores are present and, as a result, decrease the supplementary income of landowners who lease their grounds for hunting activities (Sjölander-Lindqvist 2015, Mensah et al. 2019).

Large carnivores can provide recreational benefits to hunters. Hunters can enjoy watching large carnivores or hunt them in areas where licensed hunting is allowed. When carnivore populations have attained favourable conservation status[1], licensed hunting is generally used by authorities as a management tool to control predator populations. Authorities define the maximum number of killed carnivores allowed within a county[2] for a given species in a given year. Carnivore license hunting creates tangible monetary income for landowners because hunters are willing to pay to participate in these hunts. In addition, large carnivores could indirectly benefit landowners through the potentially limiting effect of predation on ungulate population size, decreasing damage to forest and farmland (Fischer et al. 2013, Ordiz et al 2013).

Large carnivores pose noticeable economic costs at a local level that are difficult to reconcile with the more intangible biodiversity values and other ecosystem services at global scales (Farber et al. 2002). This phenomenon makes human–carnivore coexistence difficult to achieve in areas with high carnivore presence (Dickman et al. 2011, Van Eeden et al. 2018). The empirical literature has examined the economic impacts of large carnivores by estimating the resulting costs and damages on the basis of livestock depredation and reduction of game abundance (Asheim & Mysterud 2004, Bostedt & Grahn 2008, Häggmark-Svensson & Elofsson 2016, Widman & Elofsson 2018). Some researchers have studied the associated

[1] As defined by the European Commission, see http://ec.europa.eu/environment/nature/conservation/species/carnivores/conservation_status.htm#top-page. Last accessed August 7th of 2018.

[2] A county ("län" in Swedish) is an administrative territorial division in Sweden, there are 21 counties in Sweden.

livestock damage compensation schemes (Zabel et al. 2014, Persson et al. 2015, Skonhoft 2016). Several valuation studies have estimated the WTP for large carnivore conservation (Håkansson et al. 2011, Ericsson et al. 2015, Bostedt et al. 2008). Among those, Johansson et al. (2012) demonstrated that hunters have a lower WTP than others. Diverse valuation techniques have been used in research to examine hunting values of game species, including stated preferences methods such as contingent valuation (Boman et al. 2011, Zhang et al. 2004) and revealed preferences such as hedonic pricing and travel cost methods (Sarker and Surry 1998, Livengood 1983, Meilby et al. 2006, Knoche & Lupi 2007, Little & Berrens 2008, Martinez-Jauregui et al. 2015, Mensah & Elofsson 2017). A common finding in most of these studies elicits *inter alia* that trophy game and game types (i.e. species composition) are significant determinants of hunting values. However, no valuation studies have estimated the economic impact of large carnivores on hunting lease prices while simultaneously considering the effect of licensed carnivore hunting opportunities. This gap in the literature has a potentially substantial policy relevance: If hunters attach a high value to carnivore hunting, the licensed hunting could partially or completely offset the costs due to reduced availability of ungulate game species.

The purpose of this study is to estimate the effect of the presence of large carnivores on hunting leases accounting for the allotment of carnivore license hunting quotas by using the hedonic price method. A hunting lease is the payment made by hunters to landowners to rent a hunting ground and seize the harvested game; therefore, it reflects the economic value (and cost) of hunting activities. We posit that carnivore presence decreases hunting lease prices because all three carnivore species mentioned above prey on ungulate game and may, therefore, result in fewer animals available to harvest (Roe deer: Davis et al. 2016; Moose: Wikenros et al. 2015). In addition, we infer that this negative impact is reduced by the positive effect of carnivore quotas, provided that hunters receive recreational benefits from harvesting carnivores until the license quota limit is filled.

This study considers the three largest terrestrial carnivores in Sweden: the wolf, Eurasian lynx and Eurasian brown bear. We include the brown bear licensed hunting quota in our analysis because there exists a commercial market for the licensed hunting of bears in Sweden (Fischer et al. 2013), where hunting opportunities are advertised. A corresponding commercial market is not observed for lynx or wolf. An increasing number of financially motivated, guided brown bear hunts that were rarely offered in Sweden a few decades ago (Bischof et al. 2008) is resulting in an increasing number of paying foreign hunters (Swenson et al. 2017).

As hypothesized, our results provide statistical evidence that the wolf and lynx reduce hunting lease prices. Results from median regressions suggests that the (negative) marginal implicit price of a wolf territory is about SEK 1.5 million (EUR 160 thousand[3]), which is approximately 15% larger than the corresponding implicit price for a lynx territory (SEK 1.31 million or EUR 141 thousand). There is not clear evidence that bear abundance affects hunting lease prices, but as expected, licensed bear hunting yields a positive and significant effect on the lease price. Also based on median regressions, the marginal implicit price of an additional quota allotment is SEK 165 thousand (EUR 17 thousand). The ordinary least squares regressions yield larger marginal implicit prices in general. This is seemingly due to the presence of outliers in the dataset.

2. Empirical context

Hunting lease market and large carnivore species in Sweden:

According to 10§ of the Hunting Ordinance, the landowner owns the hunting right and is entitled to the wildlife seized. The payment to transfer such a right from a landowner to a group of hunters constitutes the hunting lease price. In Sweden, the approximately 300 000 hunters have invested about EUR 50 million annually in hunting (Mattson et al. 2008, Boman et al. 2011). Hunting occurs to some extent on most land where it is legally permitted, and hunting is frequently carried out by hunting teams (Mensah and Elofsson 2017). Hunting teams and lease contracts are typically very stable over time (Ericsson et al. 2010).

According to the estimations from 2015/2016 from the Swedish Environmental Protection Agency (SEPA), Sweden is home to approximately 2 800 brown bears, 1 300 lynx, and 340 wolves (Naturvårdsverket[4]). Figure A1 illustrates the population densities for wolf, lynx, and brown bear in Sweden. Moose (*Alces alces*) and roe deer (*Capreolus capreolus*) are the ungulate game species primarily affected by the wolf, lynx, and brown bear (Dahle et al. 1998, Odden et al 2006). Among the ungulates, the most hunted species in 2014/2015 were the roe deer, wild boar, and moose, with annual bags of approximately 106 000, 90 000 and 87 000, respectively (Wildlife Database[5]).

[3] Average exchange rate SEK/EUR 9.35, years 2014 and 2015. Central Bank of Sweden *(Sveriges Riksbank)*.
[4] https://www.naturvardsverket.se/Miljoarbete-i-samhallet/Miljoarbete-i-Sverige/Viltforvaltning/Rovdjursforvaltning/Inventering/. Last accessed August 27th 2020.
[5] https://rapport.viltdata.se/statistik/. Last accessed August 7th 2018.

The wolf is the largest wild canid, and occur as packs of a female–male pair with offspring, pairs without offspring or single individuals (Sillero-Zubiri et al. 2004). The wolf is mostly present in central Sweden and mainly preys on moose and, alternatively, smaller ungulates such as roe deer (Sand et al. 2016). Brown bears are omnivorous, but largely consume vegetarian food sources, and are mostly found in central and northern Sweden. In spring and summer, brown bears frequently prey on newborn moose and reindeer calves (Dahle et al. 1998). The lynx is a felid that mainly preys on medium-sized ungulates, such as roe deer, and other smaller prey (Andrén & Liberg 2015) and is found in large parts of Sweden.

After intense human persecution and lethal removal prior to and during the 20th century, the three carnivore species have recolonized large parts of their historical distribution and represent many management and conservation challenges for national and regional authorities (Laikre et al. 2003, Andrén et al. 2006, Swenson & Kindberg 2011, Skonhoft 2016, Steyaert et al. 2016).

Licensed hunting of large predators

The brown bear, wolf, and lynx are protected by the European Union Habitats Directive (Council Directive 92/43/EEC), the guiding legislation to design national and regional management plans for each large predator. Hunting of large carnivores is governed by the Swedish Hunting Act and Hunting Ordinance (1987:905), and article §23c regulates licensed hunting.

The SEPA and the County Administrative Boards (CABs) are responsible for establishing annual hunting quotas, that is, the maximum number of animals allowed to be killed each season. These numbers are determined based on predictive population growth models and population status in relation to conservation goals (Cinque 2015). Additionally, the SEPA and the CABs may allow lethal control of problem-causing individuals in counties with permanent populations of the carnivore species (Sjölander-Lindqvist et al. 2015).

Licensed hunting of carnivores allow hunters to benefit from recreational hunting of carnivores. Recreational hunting is used by authorities as a cost-efficient strategy to limit population growth or size, mitigating direct economic losses caused by wildlife damage (Bischof et al. 2012). Recreational hunting also serves as a mechanism to increase local involvement in carnivore management and, as result, enhance acceptance of large predators among hunters (Sjölander-Lindqvist 2015). Moreover, the provision of and demand for guided bear hunts in Sweden reveal the existence of a bear hunting market that has expanded in recent years (Bischof et al. 2008, Fischer et al. 2013, Swenson et al. 2017).

Carnivore–human conflicts because of predation on ungulate game populations is expected to be a pivotal factor that negatively affect hunting lease. Given that licensed carnivore hunting is highly restricted, the possibility of license hunting might not outweigh the negative impact of larger abundance of carnivores, but could at least moderate the effect. In addition, it should be recognized that if landowners or hunters highly value the ecological contribution of predators as keystone species adding services to the local ecosystem, this would further ameliorate the negative impact on hunting lease prices due to predation of ungulate game species. In the following, this study provides additional details on the ultimate effect of large carnivores on lease prices.

3. Hedonic model

In a competitive market, the equilibrium price of a composite good is determined by the interaction of sellers and buyers. For the market to be competitive, it must be sufficiently large for the buyers and sellers to be price takers. In such a market, different product characteristics affect the price of the composite good. The hedonic pricing method, which consists of regressing the prices of a differentiated good on the quantities of various characteristics, permits identification of the impact of different attributes on the equilibrium price, which is attained upon convergence of the bid and offer functions of the sellers and buyers (Rosen 1974). The price of the composite good is then interpreted as a function of a vector of product attributes, and the partial derivative with respect to each attribute denotes its marginal implicit price (Taylor 2003).

The hedonic price theory can be applied to the wildlife arena to decompose the values of a hunting ground into its constituent attributes. One of the most important attributes is the harvest of ungulate game. Ungulate game can provide recreational benefits to hunters derived from hunting and viewing, and harvesting and viewing opportunities are increasing in the population of the game species. However, ungulate game also entail substantial costs to landowners due to browsing damages and overgrazing, which can reduce forest and agriculture productivity (Côté et al. 2004). Hunters could reduce these damages by harvesting the game, but would only be willing to reduce the population to a certain extent, as they would want to maintain a relatively large ungulate populations to secure the possibilities for substantial future harvests. Thus a high harvest, which can be sustained over several years, requires a higher ungulate population and is therefore associated with a higher cost for the landowner. Admittedly, a landowner could choose to rent out the land on a short-term basis in order to substantially reduce ungulate game population. However, such short-term contracts where hunters substantially reduce the game populations are very close to non-existent[6]. Instead hunting lease contracts and hunting teams are typically stable over time (Ericsson et al. 2010). This can be explained by the role of social networks for the uptake of hunting team members and the importance of trust between the landowner and the hunting team (Mensah and Elofsson 2017). An additional likely reason for the absence of such contracts is hunters' ethical concerns regarding such substantial reductions in the populations. Moreover, landowners may derive recreational benefits from the presence of ungulates on their land, and the opportunity of viewing those, implying that they may not benefit from a reduction of game populations to very low levels. Hence, the benefits and costs from ungulate harvesting and populations accrued by hunters and landowners are important determinants of the bid and offer functions, respectively. The bid price resulting from hunter's utility maximization represents the maximum pecuniary amount that he or she is willing to pay for renting the hunting plot, while the offer price derived from the landowner's net profit maximization is the minimum amount that he or she is willing to accept to lease out the hunting land. The equilibrium market price is determined by the convergence of bid and offer prices (Palmquist 1989). The literature shows that the equilibrium price for hunting opportunities is a function of the hunting ground characteristics, such as ungulate game harvests, hunting experience, wildlife population densities and intrinsic features of the landowner and the hunter (Pope & Stoll 1985, Messonier & Luzar 1990, Meilby et al. 2006, Zhang et al. 2006, Lundhede et al. 2015, Mensah and Elofsson 2017). The specific attributes addressed in this study are based on previous empirical studies and are described in detail below.

Because of the large size of the Swedish hunting lease market, landowners (sellers) and hunters (consumers) are assumed to be price takers. Given the availability on the market of both long and short-term hunting leases, where the latter are adverted in various internet fora, we assume that restrictions to enter the market are negligible.

In particular, large carnivore presence can be seen as an environmental attribute of hunting lease prices as it entails costs to hunters due to predation of ungulate game, yet it can also bring benefits to landowners (e.g., private forest owners or forest companies) associated with lower herbivore density. We use a hedonic pricing analysis to address the impact of large carnivores

[6] There have been attempts from the Swedish Hunters' Association to introduce such contracts in order to reduce the substantial damages from wildboar, the so called Wild Boar Asssistant Scheme, but the interest from landowners has been minimal.

on hunting lease prices. By regressing lease prices on the set of constituent characteristics, we can estimate the marginal implicit price of large carnivore presence. We formulate our hedonic price function as follows:

$$lnP = \alpha + \boldsymbol{\beta H} + \boldsymbol{\gamma C} + \phi Q + \boldsymbol{\delta A} + \boldsymbol{\theta G} + \varepsilon, \qquad (1)$$

where P is the lease price per hectare paid annually by a hunting team. Thus, the leaseholder is the hunting team, which constitutes of several hunters that together share and pay the costs for the lease. Hedonic pricing theory does not pose any constraints on the choice of functional form (Cropper et al. 1988, LeGoffe 2000, Meilby et al. 2006), but the semi-logarithmic specification of the dependent variable used in our study has been extensively used in preceding hedonic literature addressing hunting values (Livengood 1983, Hussain et al. 2007, Munn & Hussain 2010, Mingie et al. 2017). There are two major motives for our choice of functional form: to reduce the impact of outliers in the dependent variable by applying a natural log-transformation, and to be able to provide a comprehensible economic interpretation of the marginal attribute prices. Alternative specifications, such as Box-Cox transformations, would make marginal implicit prices very difficult to interpret given how some of our (ecological) explanatory variables are constructed.

The variable $\boldsymbol{C}$ is a vector of indexes measuring carnivore abundance for the wolf, lynx and brown bear, respectively. All three species prey on ungulate game (e.g. moose and deer) and may therefore influence game population sizes, which could result in fewer animals available for hunting. Moreover, fear of large carnivores, as well as attacks of wolves and brown bears on humans, livestock, and hunting dogs have been documented (Frank et al. 2015) and may have a negative effect on hunting lease prices. Both effects are conducive to predict a negative relationship between carnivore presence and lease price.

The variable Q is the licensed bear hunting quota set by the respective CABs in 2014. We expect that larger quotas are associated with higher lease prices because bear hunting may provide recreational benefits to hunters. In 2014, no quotas were allotted to wolves, and only four counties allotted quotas to lynx (Table 1). This information dissuades us from including the wolf and lynx quotas in the model due to an insufficient number of non-zero observations. Conversely, brown bear quotas provide reasonable variability to our estimations because one third of the counties allotted bear quotas in 2014[7].

[7] 42.7% of the observations in this regressor have non-zero values.

The variable $\boldsymbol{A}$ is a vector of hunting team and hunting ground attributes, that is, hunting ground size and number of teammates. Hunting ground size has been found to affect lease prices significantly, although with mixed signs for different studies. Mensah & Elofsson (2017) argue that leases respond positively to larger areas because they increase the probability of a successful hunt; however, Lundhede et al. (2015) found a negative relationship and state that larger grounds increase the effort and marginal cost to harvest game, which decreases the benefits from hunting. The number of teammates is included in this vector as a measure of congestion (Livengood 1983). We hypothesize a positive sign for this regressor because larger hunting teams may result in potential damage to and deterioration of the hunting ground; therefore, a higher payment is demanded by landowners. Finally, land ownership is a dummy variable equal to 1 when a forest company owns the hunting ground. Forest companies (public and private) own 42% of the forested territory and have been found to charge an overhead percentage above the regular leasing fees (Bergman & Åkerberg 2006). In this regard, we intend to determine whether lease prices differ depending on land ownership.

The variable $\boldsymbol{G}$ is a vector of municipality attributes that potentially influence lease prices, such as human population density, municipality income per capita, and distance to the nearest big city. Population density and income per capita control, respectively, for demographic and structural differences between municipalities. We expected these variables to positively affect the demand for hunting and, hence, the hunting lease price. According to the literature, demand is lower for hunting grounds located farther from urban areas (Little and Berrens 2008); therefore, we also include distance to the nearest big city (Stockholm, Gothenburg or Malmö) and predict a negative sign. This expected negative relationship could be explained by the travel cost incurred by urban hunters to reach remote hunting sites: hunters' willingness to pay (WTP) may decrease for (overly) distant hunting areas.

In Eq. (1), $\boldsymbol{H}$ is a vector of proxies for ungulate game harvest. Game harvest is a core constituent attribute that provides benefits to hunters for the shooting or the meat value (Lundhede et al. 2015, Mensah & Elofsson 2017). A particular challenge when studying the impact of carnivores on hunting lease prices is that one can expect that, to a considerable extent, the impact occurs through the effect of carnivores on the harvest of ungulate game. The potential harvest of game at a given location depends on the habitat, climatic conditions, and abundance of carnivores. To detach carnivore effects from other determinants of hunting leases, we use snow depth and forest productivity as proxy variables for the potential ungulate game harvest in the absence of carnivores, which is motivated as follows. First, snow depth and forest productivity are key factors affecting ungulate game behaviour, habitat selection, and hence, the spatial distribution of game populations (Mysterud et al. 1997, Lundmark & Ball 2008). Hence, habitat productivity and climate are indicators of the potential density of ungulate game populations that could occur in the absence of carnivores. Second, ungulate game population size is typically strongly correlated with harvests, which is illustrated by the frequent use of harvests as proxies for population sizes (Elton & Nicholson 1942, Cattadori et al. 2003).

Snow depth could affect wildlife abundance at a given location in the short term and long term. Moose and deer species generally migrate to shallower snow depths in winter because deep snow reduces available forage and eventually hampers free mobility (Mysterud & Østbye 2006). Moreover, harsh winters can affect reproduction and survival of wintering ungulates (Gaillard et al. 1993, Kjellander & Nordström 2003, Lundmark & Ball 2008). Mech et al. (1987) shows that the current winter's snow and the severity and persistence of previous winters can influence moose and deer populations. Thus, snow depth can provide a good measure of winter severity and is here used to control for the effect of winter conditions on the population dynamics of moose and deer species. Because the hunting season 2014/2015 ends in January 2015 for the moose and roe deer, we define our proxy variable as the average snow depth over the winter months: December 2013 to March 2014, and December 2014 to January 2015. In this manner, we include all winter months in the year before 2014/2015 and during 2014/2015, which is motivated by the potential importance of both for game harvests in 2014/2015[8].

The forest and vegetation structure are regulating mechanisms for herbivore densities that may affect their population growth rates (Hjeljord et al. 1990). In general, moose have been observed to select high-productivity forests over low-productivity forests (Bjørneraas et al. 2012), and fertile soils usually provide more suitable habitats for wildlife than poorer sites (Meilby et al. 2006). To that extent, the productivity of forest land is associated with moose distribution and may serve as a good proxy for ungulates' abundance. Accordingly, the habitat types of high-productivity forests increase the abundance of ungulate game and, as a result, increase hunters' WTP. We introduce forest productivity in the model as the mean annual volume increment of forest trees per hectare (m^3/ha)[9]. We take the average of the mean volume

[8] Snow depth may have a more long term effect on ungulates behavior than a short term effect; therefore, it would have been better to use a longer time span (e.g. winter months of the last 5–10 years). However, snow depth data is available for only 72 municipalities out of the 154 of our sample, and retrieving data for the previous 5 to10 years would purport further data loss.

[9] More specifically, we consider the forest land suitable for forest production, also defined as *productive forest land*. Tree species include pine, lodgepole pine, spruce, birch, oak, beech, and other broadleaves.

increments of the preceding year (2013) and the year of the hunting season (2014) to account for any lagged or lingering effect of forest productivity on moose abundance.

Conceptually, the use of snow depth and forest productivity in the model allows us to exclude harvest from the hedonic model without having an omitted variable bias. As argued, we expect a high correlation between the two proxies and the ungulate game harvest that could be obtained in the absence of carnivores, and consequently, any remaining bias is substantially reduced.

Marginal implicit prices

As asserted throughout, this paper is predominantly interested in examining and quantifying carnivore effects on hunting lease price. To calculate the marginal implicit price of each carnivore, we rewrite Eq. (1) in exponential form:

$$P = e^{\alpha+\boldsymbol{\beta H}+\boldsymbol{\gamma C}+\phi Q+\boldsymbol{\delta A}+\boldsymbol{\theta G}+\varepsilon} \qquad (2)$$

Next, we differentiate this hedonic function with respect to the abundance of each carnivore species (C). Hence, the marginal implicit price of species $k =$ {wolf, lynx, bear} is given by:

$$\frac{\partial P}{\partial C_k} = \gamma_k\left[e^{\alpha+\boldsymbol{\beta H}+\boldsymbol{\gamma C}+\phi Q+\boldsymbol{\delta A}+\boldsymbol{\theta G}+\varepsilon}\right] = \gamma_k P \qquad (3)$$

Bear quotas are determined as a percentage of the estimated bear population size within each county (Bischof & Swenson 2009). We assume that bear quotas are exogenous (Q) to the landowners and hunters, i.e. the market agents are not able to influence the size of the quota. This is motivated by the fact that the quota is determined by the CAB, which regulates the population based on the criteria mentioned in Section 2, using historical data on the bear population and its development. This implies that we can calculate the marginal implicit price of the bear quota by differentiating the hedonic function (written as in Eq. (2)) with respect to the quota regressor:

$$\frac{\partial P}{\partial Q} = \phi\left[e^{\alpha+\boldsymbol{\beta H}+\boldsymbol{\gamma C}+\phi Q+\boldsymbol{\delta A}+\boldsymbol{\theta G}+\varepsilon}\right] = \phi P \qquad (4)$$

4. Data

We surveyed a random selection of 2014 individuals with a national hunting license registered in the official Hunters Registry database managed by the SEPA. The overall purpose of the survey was to map hunters' attitudes to hunting ethics and illegal hunting. It included questions on the respondents' hunting motives and experience, and hunting practice. It further asked questions on attitudes to wildlife management and policy and wolf conservation. A pilot was carried out with thirty hunters with varying background and in different parts of the country, and the survey was adjusted based on the responses obtained. The survey was first sent out by mail in May 2016 and was followed by two reminders. To encourage participation, a hunting trip worth SEK 7 500 was drawn out among the respondents. The questionnaire was responded to by 957 hunters (i.e. 47.5% response rate), out of which 314 answered the questions used for this study. From the survey, we only used variables for the hunting lease price[10], the size of the hunting ground, the number of team members, and whether the land was owned by a forest company. Concerning the hunting lease price, the survey asked hunters the price per hectare (P) that they actually pay to lease the right to hunt in their most frequented hunting site for the 2014/2015 season. Hunting plot areas reported in the survey are located in 154 Swedish municipalities out of a total of 290 across all 21 counties (Fig. A2 in the Appendix). The female rate and the average age in our sample are 3.14% and 56.9 years, respectively, compared with 6.28% and 52.6 years in the Hunters Registry. In addition, 2.22% of the surveyed hunters did not specify gender or age. The modest response rate of 47.5% could be explained by the comparatively sensitive nature of the questions posed on individual hunters' attitudes. Further details on the survey, as well as the questionnaire itself, can be found in Peterson et al. (2019).

One limitation of survey-based studies is that they may suffer from self-selection bias. For example, hunters answering the questionnaire could have different characteristics and motivations compared with hunters who do not participate in the survey. However, for a competitive market where hunters are price takers, such as assumed here, the estimates are unlikely to be influenced by sampling bias (e.g., Lundhede et al. 2015).

Provided the judgement of experienced hunters, we considered outliers to be leases with prices greater than SEK 1 000 (EUR 93.5) per hectare. There were five answers beyond this limit, which we set to the 98th percentile of the data, that is, we performed a 98% winsorization to reduce the effect of outliers (Ruppert 2006, Sandkild 2010). Additionally, size of hunting area and number of team members elicit a pronounced long-tail (left-skewed) distribution, suggesting the presence of outliers in the answers[11]. All three variables are transformed with a

[10] There is no official registration of hunting lease contracts or hunting lease prices, and therefore it is necessary to collect such data through surveys.
[11] Pinpointing the reason of these outliers in survey research is extremely difficult, for example, misreading or misinterpreting the question, uneducated guesses, and misguided or erroneous information.

natural logarithm to decrease data variability and reduce the influence of outlying observations (Figs. A3–A8 in the Appendix).

Wolf, lynx, and brown bear presence are measured, respectively, as an index of density of each carnivore in every municipality for the years 2013/2014. Carnivore data was collected from the Scandinavian database of large carnivore surveys (www.rovbase.no), which is a management tool for monitoring carnivores populations in Norway and Sweden. The brown bear index is based on identification of bear individuals from scat surveys (DNA inventories) and reflects the relative bear density in each municipality (Kindberg et al. 2011).

The main monitoring units are wolf family groups (i.e. a pair with offspring) and pairs (i.e. territorial [scent-marking] pairs), and lynx family groups[12] (i.e. a female with young of the year). The monitoring data comprises coordinates for the centre point of all observations documented within the monitoring programme assumed to belong to the same family group or pair (Anon. 2014, Svensson et al. 2014). We created a species-specific buffer zone around each centre point, corresponding to published home range sizes for the wolf and lynx, 1 000 km^2 for wolves (Mattisson et al. 2013) and 320 km^2 for lynx (Aronsson et al. 2016).[13] To obtain a relevant index reflecting lynx and wolf presence in each municipality, we calculated the area of all buffer zones for each species overlapping each municipality and divided by the area of the municipality (see Appendix). We allowed the buffer zones to overlap, to represent a more realistic abundance of the two carnivore species within each municipality; hence, the index for wolves ranged from 0 to 2.29, and the index for lynx ranged from 0 to 0.77.

Further, the bear quotas by county for 2014 are documented in the website of the National Veterinary Institute (Table 1). In 2014, the total allotted quotas to the brown bear, lynx, and wolf were 273, 30, and 0, respectively. Considering that quotas are allocated at the county level and some counties are considerably larger than others, we control for the size of the counties in relation to the size of the hunting plot. Therefore, the explanatory variable Q of Eq. (3) is defined as the bear hunting quota per county multiplied by the ratio of hunting plot area to county area.

[12] This means that one monitored lynx family group represents approximately 5.48± 0.40 individuals as well as unmonitored subadults, adult males, and barren females (Andrén et al. 2002).

[13] This method has limitations because it is unknown how well the assumed center points of observations during the monitoring represent the actual location of home ranges. Further, the species-specific buffer zones around these are assumed to be circular and, hence, do not consider the natural variation in home range shape and size. Despite these limitations, we argue that our index is the best possible available for this study.

Table 1. Hunting quotas for brown bear and lynx in 2014 for counties that issued quotas for one or both species*

County	Brown bear	Lynx
Värmland	2	0
Dalarna	62	0
Gävleborg	36	6
Västernorrland	21	7
Jämtland	60	0
Västerbotten	25	6
Norrbotten	67	11
Total	**273**	**30**

Notes: No quotas were allotted to the wolf in 2014

Source: Statens Veterinärmedicinska Anstalt, SVA 2017.

Income per capita, population density, and distance from the hunting site to the nearest big city are variables constructed per municipality by retrieving information from Statistics Sweden. Income per capita is measured as the gross average pre-tax income earned in each municipality by local individuals older than 20 years in 2014 real prices (Swedish kronor); population density is the number of residents per squared kilometre in 2014, and the nearest big city is the distance in kilometres from the municipality where the hunting plot is located to the most proximal big city: Stockholm, Gothenburg, or Malmö. To decrease the high dispersion of these three variables, we transform them with a natural logarithm.

Finally, snow depth is extracted from the Swedish Meteorological and Hydrological Institute, and forest productivity is obtained from the Swedish National Forest Inventory (Riksskogstaxeringen, SLU).

Table 2. Descriptive statistics *(n = 323)*

Variable	Description	Mean	Std. Dev.	Minimum	Median	Maximum
Hunting lease price	Hunting lease price (SEK/ha)	96.68	168.19	0.17	50	1 000
log(price)	Log-transformed hunting lease price	3.49	1.71	-1.79	3.91	6.91
Area	Size of hunting area (hectares, ha)	2 727	6 401.84	4.00	1 005	80 000
log(area)	Log-transformed size of hunting area	6.89	1.54	1.39	6.91	11.29
Members	Number of hunting team members	18.03	28.39	1	11	350
log(members)	Log-transformed number of members per hunting team	2.55	0.80	0.69	2.48	5.86
Forest_prod	Forest productivity (m^3/ha)	6.21	1.62	2.80	6.80	8.30
Snow_depth	Snow depth (m)	0.11	0.14	0	0.05	0.60

Wolf index	Wolf index	0.20	0.45	0	0	2.29
Lynx index	Lynx index	0.16	0.16	0	0.12	0.77
Bear index	Bear index	0.12	0.24	0	0.16	1.73
Bear_quota	Bear quota	17.72	25.88	0	36	67
Bear_quota_adj	Bear quota adjusted by the size of the hunting ground with respect to the size of the county	2.87	3.25	0	0	1.76
Popdens	Municipality population density (population per km^2)	31.16	32.68	0.20	19.80	176.70
log(popdens)	Log-transformed municipality population density	3.44	3.49	-1.61	2.99	5.17
Income_pc	Municipality income per capita (SEK 1 000)	51.28	218.86	1.24	13.47	2 415.25
log(income_pc)	Log-transformed municipality income per capita	3.94	5.39	0.22	2.60	7.79
Distance	Distance to the nearest big city (km)	256.68	216.27	21.29	183.98	942.69
log(distance)	Log-transformed distance to the nearest big city	5.55	5.38	3.06	5.21	6.85
Company	Land ownership (dummy=1 for forest company)	0.26	0.44	0	0	1
Municipality area	Size of the municipality area of our sample (ha)	214 500	2 886	185	114 082	1 688 667
County area	Size of the county area (ha)	2 523 855	2 579 944	294 100	1 117 100	9 891 100

5. Results and discussion

We estimate the hedonic function by using ordinary least squares (OLS) and least absolute deviation (LAD) regressions. As asserted in the data section, the occurrence of outliers in the survey dataset is a challenge, and may influence the *t* values and hence the significance of the estimates in OLS regressions. Anomalous observations in a homoskedastic model can potentially make the model heteroskedastic (Gujarati & Porter 2009, Alih & Ong 2015) and may produce long-tail (skewed) distributions. Hence, we address this problem by implementing the LAD method, i.e. median regressions, and compare the results with those of OLS. Least absolute deviation estimations can provide a richer characterization of our data and produce estimates more robust to outliers (Cameron & Trivedi 2010).

Compared to OLS, median regressions are more resistant to outliers because the estimates minimize the sum of absolute residuals instead of the sum of squared residuals. LAD coefficients are more efficient than least squares estimates due to the presence of outliers in our dataset and notably skewed distributions (Bassett & Koenker 1978). Despite this statistical desirability of using median estimations in our analysis, it is worth noting some drawbacks. LAD is a maximum likelihood estimator and asymptotically unbiased only if the disturbances follow a Laplace distribution (Narula & Wellington 1977). Moreover, LAD regressions can produce multiple solutions and exhibit instability at datasets that are far from collinear (Ellis 1998). We make use of the principal component analysis as an attempt to inspect the latter problem. The principal component analysis method might be employed as a remedial procedure of multicollinearity (Willis & Perlack 1978). Thus, we address whether LAD estimates are too sensitive upon removal (or addition) of covariates after the model has been estimated with principal components.

We estimate our hedonic function with both OLS and LAD using different types of standard errors. We start with conventional standard errors, however the heteroskedasticity tests of Breusch-Pagan, White, and Koenker-Basset reject the null hypothesis of homoskedastic residuals. Hence, we proceed by estimating the model using robust standard errors. The Jarque-Bera and Shapiro-Wilks tests reject the null hypothesis of normally distributed residuals, and this may be explained by the skewed distribution of some key variables in the model (despite transforming these variables with a natural logarithm as discussed in the data section; Figures A3–A8 in the Appendix). As a result, we also estimate the hedonic function using bootstrapped standard errors (5 000 replications), which mainly rely on the empirical distribution of our sample instead of asymptotic normality. Furthermore, we cluster standard errors on a municipality level to account for any unobserved correlation within municipalities. Two unobserved attributes are considered in this regard: the presence of hunting clubs is a factor potentially affecting hunting leases (Livengood 1983, Mingie et al. 2017); and local landscape characteristics could influence hunting rental prices (Meilby et al 2006). Ignoring error correlation within clusters (i.e. within municipalities) can lead to deceptively small standard errors and, thus, a great loss of efficiency in OLS estimations (Cameron & Miller 2015). We only report the model with clustered standard errors in Table 3 provided the latter empirical justifications and because the statistical significance of the regressors does not differ sharply with the use of different standard error types (i.e. conventional, robust, bootstrapped and clustered).

Table 3. OLS and LAD estimations of the hedonic price function with standard errors clustered on the municipality level.

Dependent variable: *log (price)* *Price: Hunting lease price per hectare (for the hunting season 2014/2015)*	**OLS**	**LAD**
log(area)	- 0.34 *** (0.0701)	- 0.21 *** (0.0578)
log(members)	0.08 (0.1091)	0.09 (0.0656)
Forest_prod	0.21 ** (0.1065)	0.39 *** (0.0837)

Snow_depth	- 0.15	- 1.32
	(1.0883)	(1.0094)
Wolf index	- 0.43 *	- 0.39 *
	(0.2242)	(0.2205)
Lynx index	- 1.37 ***	- 1.07 ***
	(0.4964)	(0.3648)
Bear index	- 0.72 *	- 0.09
	(0.4313)	(0.3284)
Bear_quota (adj)	2.25 ***	1.58 ***
	(0.6009)	(0.2776)
log(pop_density)	- 0.08	0.005
	(0.1138)	(0.0926)
log(income_pc)	- 0.06	- 0.05
	(0.0813)	(0.0682)
log(distance)	- 0.42 ***	- 0.19
	(0.1438)	(0.1219)
Company	- 0.23	- 0.004
	(0.1939)	(0.1306)
Intercept	7.34 ***	3.93 ***
	(1.5105)	(1.2254)
R-squared:	0.4398	

*Significance levels: *** 1%, ** 5%, * 10%*
Clustered standard errors on the municipality level (shown in parentheses)

Results show that carnivore abundance yields negative estimates as expected. The lynx estimate remains always significant at 1% and thus displays the most robust result among the three carnivore coefficients. The wolf estimate is significant at 10% level, and so is the case of the bear index in the OLS regression. The bear license-hunting quota is positive and significant at 1% in the OLS and LAD estimations, despite the small variability of the bear quota regressor[14].

As was expected, forest productivity is positive and significant. Snow depth and human population density are insignificant in the regressions; however, this outcome could be due to the presence of some degree of multicollinearity within the models. Tables A3 and A4 (in the

[14] Only seven counties (out of the 21) allocate quotas to bears, which results in having several observations equal to zero for this variable (i.e. 57% of the total number of observations correspond to a county where no quotas were allotted).

Appendix) show evidence of high correlation, especially among the four variables: forest productivity, snow depth, human population density, and distance to nearest big city. However, the exclusion of any of these four variables would potentially bias our estimates because of their ample empirical relevance (the Ramsey RESET test fails to reject the null hypothesis of correct specification). Forest productivity has a variance inflation factor (VIF) of 4.27, and a correlation of -81% with snow depth. Similarly, population density shows a VIF of 4.39, and a correlation of -63% with snow depth, and -71% with distance to closest big city. Moreover, the variance inflation factor of all variables never exceed 10, which has been suggested as a guideline to detect serious multicollinearity in numerous works (reviewed in O'Brien 2007). However, to check the severity of and circumvent possible multicollinearity problems, we also estimated the hedonic function using a principal component analysis. The first and second components explain 90% of the variance of the four variables; thus, we included these two components in the regressions (Tables A1 and A2 in the Appendix). Yet, results are similar in terms of the sign and significance of the point estimates obtained in Table 3 (see also Columns 2-4 of Table A2 in the Appendix).

The number of hunting team members is insignificant in all estimations, which suggests that size of the team does not affect the lease price. On the other hand, size of the hunting ground yields a negative and highly significant point estimate. If instead, we included size of hunting ground per hunting team member as one explanatory variable, the resulting coefficient for this variable would still be negative and significant for OLS (-0.27***) and LAD (-0.15**). These results show that the lease price decreases with larger areas (or with larger areas per team member), which validates the findings of previous studies. Shrestha and Alavalapati (2004) attribute the negative sign to the diminishing marginal returns to scale, while Lundhede et al. (2015) state that greater plots entail higher hunting effort at possibly higher marginal costs to the landowner.

The distance to the nearest big city is negative and significant in the OLS case, which corroborates the findings in the literature. Proximity to urban areas decreases the travel time of urban hunters, increasing the demand for hunting grounds close to big cities. In northern Sweden the Sami communities have hunting rights within the reindeer herding areas, which could potentially affect lease prices. We added a dummy variable in the hedonic model to control for municipalities with Sami administrative communities. Because the inclusion of this dummy was not statistically significant and did not alter the results, we excluded the variable from the final estimations.

Marginal implicit prices of large carnivores:

We use the point estimates (γ_k) of OLS and LAD regressions respectively, in order to calculate and compare the marginal implicit price of each carnivore species abundance. To compute the marginal implicit prices that follow from the OLS regressions we use the average lease price of the sample, i.e., $P = 96.68$, and to calculate the marginal implicit prices derived from the LAD estimations we use the median lease price, which is $P = 50$ (Table 2). This choice is made because the LAD method models the conditional median, and therefore the median (price) is a more suitable measure of central location for the dependent variable (Hao & Naiman 2011), while the OLS models the conditional mean, motivating the use of the average price in the calculations.

In the regressions, carnivore densities are measured through indexes. A unit increase in the index would, for example occur if a municipality with a zero population of the carnivore would receive carnivores of a sufficient number for the territories to cover all land in the whole municipality. This is a large change in carnivore abundance. To have marginal implicit prices that are intuitive and policy relevant, we chose instead to compute the marginal implicit price of an additional carnivore territory (for the wolf and lynx) and an additional individual (for the brown bear), details on these calculations are available in the Appendix. All marginal implicit prices are presented in both SEK and EUR.

<u>Lynx and wolf:</u>

In a municipality of average size (214 500 hectares), one additional lynx buffer area implies an increase in the lynx index by 0.1492 units[15]. This size of a change in the lynx index implies a marginal implicit price equal to 7.98 SEK/ha (0.85 EUR/ha) for the LAD case, and 19.72 SEK/ha (2.1 EUR/ha) for the OLS case. Next, one can first note that our model expresses the impact of an increase in lynx abundance in a municipality on hunting lease prices in the same municipality, without requiring any overlap between the lynx home range and the affected hunting ground. Hence, the effect of an additional lynx buffer area on hunting lease prices occurs on all hunting land in the municipality. To obtain the marginal implicit price of one lynx buffer, we therefore multiplied the marginal implicit price per hectare by the average total area of hunting land in the municipalities in our sample. This exercise yields a marginal implicit

[15] The lynx buffer area can be found in the Data section.

price of SEK 1.31 million per lynx buffer (EUR 141 thousand) for the LAD estimate, and SEK 3.26 million (EUR 349 thousand) for the OLS estimate (Table 4).

Knowing that the wolf and lynx indexes have a similar construction, differing only in the buffer sizes (see Data section), the procedure to calculate the marginal implicit prices of wolf presence is analogous to that of the lynx. One additional wolf buffer implies an increase in the wolf index corresponding to 0.4662 units. For the LAD estimations, one additional wolf buffer decreases hunting leases by SEK 9.09 (EUR 0.97) per hectare in the affected municipality, equivalent to SEK 1.5 million per wolf buffer (EUR 160 thousand). For OLS, an additional wolf buffer area reduces the lease price by SEK 19.38 (EUR 2.07) per hectare, i.e. SEK 3.2 million (EUR 342 thousand) in a municipality.

<u>Brown bear:</u>

The brown bear index is calculated in a different way than the indexes for lynx and wolf. A couple of different steps were therefore required to obtain the marginal implicit price. First, we converted the brown bear density index to units of individual brown bears. We perform that task by comparing bear density in terms of numbers per municipality with the index used in the regressions. The number of brown bears per municipality was obtained by dividing the predicted numbers of brown bear in 2013 (Swenson et al. 2017), by the average municipality area in our sample, thereby obtaining on average 14.65 brown bear individuals per municipality. We assume that this corresponds to the average of the brown bear index in our dataset, equal to 0.12, and that brown bear numbers and the index are linearly related, see Appendix. An additional brown bear in a municipality then corresponds to an increase in the brown bear index by 0.0084 units. An increase in the bear density index by 0.0084 units implies a decrease in hunting lease price by 0.03 SEK/ha (0.003 EUR/ha) according to LAD regressions and by 0.58 SEK/ha (0.06 EUR/ha) for OLS. Following the same procedure as for lynx and wolf, i.e. summing the marginal implicit price per hectare over all hunting land in an average municipality, the marginal implicit price per brown bear individual equals SEK 6.2 thousand (EUR 668) according to LAD estimates and SEK 96 thousand (EUR 10 thousand) by OLS. It is important to note that for the case of the brown bear, the marginal implicit price from the LAD regression is not statistically significant (Table 3), whereas from OLS it yields 10% significance level. Table 4 reports the marginal implicit prices described in this subsection with corresponding confidence intervals.

Table 4. Marginal implicit prices for the three carnivore species in Sweden 2014/2015.

Marginal implicit price of an additional lynx territory	OLS	95% confidence interval (OLS)		LAD	95% confidence interval (LAD)	
		Lower bound	Upper bound		Lower bound	Upper bound
Hunting ground (SEK/ha)	-19.72	-33.80	-5.73	-7.98	-13.32	-2.65
Hunting ground (EUR/ha)	-2.1	-3.61	-0.61	-0.85	-1.42	-0.28
Municipality (million SEK)	-3.26	-5.58	-0.94	-1.31	-2.20	-0.43
Municipality (thousand EUR)	-349	-596	-100	-141	-235	-46.8

Marginal implicit price of an additional wolf territory	OLS	90% confidence interval (OLS)		LAD	90% confidence interval (LAD)	
		Lower bound	Upper bound		Lower bound	Upper bound
Hunting ground (SEK/ha)	-19.38	-36.05	-2.71	-9.09	-17.57	-0.61
Hunting ground (EUR/ha)	-2.07	-3.86	-0.29	-0.97	-1.88	-0.07
Municipality (million SEK)	-3.2	-5.95	-0.44	-1.5	-2.9	-0.1
Municipality (thousand EUR)	-342	-636	-47	-160	-310	-10.7

Marginal implicit price of an additional bear individual	OLS	90% confidence interval (OLS)		LAD
		Lower bound	Upper bound	
Hunting ground (SEK/ha)	-0.58	-1.16	-0.0068	-0.03
Hunting ground (EUR/ha)	-0.06	-0.12	-0.0007	-0.003
Municipality (million SEK)	-0.096	-0.19	-0.0012	-0.006
Municipality (thousand EUR)	-10.26	-20.53	-0.1283	-0.668

Notes: Prices as of year 2015. Average annual exchange rate SEK/EUR 9.35 according to the Central Bank of Sweden (Sveriges Riksbank).

Based on the significance level of each carnivore index coefficient (Table 3), we compute a 95% confidence interval for the marginal implicit price of lynx and a 90% confidence interval for wolf and bear. The non-significance of the bear index coefficient yielded from the LAD method prevent us from calculating the corresponding confidence interval for the marginal implicit price of an additional bear individual.

We test if the LAD point estimates are statistically different from the OLS point estimates (Table 3) in order to determine whether the marginal implicit prices significantly differ from each other when comparing the two methods. By implementing a pairwise Z equality test (Clogg et al. 1995, Paternoster et al. 1998), we fail to reject the null hypothesis that each LAD point estimate is statistically different from the corresponding OLS point estimate[16]. Namely, the marginal implicit prices of each carnivore species are not statistically different when comparing the LAD and the OLS calculations[17]. Nonetheless, we take the marginal implicit

[16] The test is defined as $Z = \frac{\gamma_{LAD} - \gamma_{OLS}}{\sqrt{SE(\gamma_{LAD})^2 + SE(\gamma_{OLS})^2}}$, where SE is the standard deviation of the γ coefficient. We implement the test for each carnivore index and the bear quota. In all cases, we fail to reject that the coefficients are equal when comparing LAD and OLS.

[17] The point estimates are not statistically different; however, the marginal implicit prices reported in Table 4 seem to differ considerably because of the procedure to calculate these. As previously stated in this section,

prices from LAD as reference values for policy considerations and for comparing with other studies in the economic literature. This is primarily because of the reasons already cited about the relatively higher efficiency and robustness of the LAD method in the presence of outliers in our dataset.

Marginal implicit price of regulated bear hunting:

We calculated the marginal implicit price of the bear quota using Eq. (4). When constructing the bear quota variable, it was assumed that the quota was proportionally distributed across all hunting land in the county (cf. Data section), and the calculation of the marginal implicit price of the bear quota took this into account. The computations were as follows: for the average size of a hunting ground (2 727 hectares) and the average size of a county (2 523 855 hectares), an increase in one bear quota allotment increases lease prices by 0.08 SEK/ha (0.008 EUR/ha) for the LAD case, and by 0.23 SEK/ha (0.02 EUR/ha) for OLS. We sum this result over the whole county and obtain a marginal implicit price of SEK 165 thousand (EUR 17 thousand) and SEK 456 thousand (EUR 48 thousand), respectively (Table 5). Details on the calculations can be found in the Appendix.

Table 5. Marginal implicit price for licensed bear hunting in Sweden 2014/2015.

Marginal implicit price of an additional bear quota allotment	OLS	95% confidence interval (OLS)		LAD	95% confidence interval (LAD)	
		Lower bound	Upper bound		Lower bound	Upper bound
Hunting ground (SEK/ha)	0.23	0.112	0.358	0.08	0.056	0.115
Hunting ground (EUR/ha)	0.02	0.012	0.038	0.008	0.006	0.012
Municipality (million SEK)	0.456	0.217	0.695	0.165	0.108	0.223
Municipality (thousand EUR)	48	23.21	74.33	17	11.55	23.85

Notes: Prices of 2015. Average annual exchange rate SEK/EUR 9.35 according to the Central Bank of Sweden (Sveriges Riksbank).

We compute a 95% confidence interval for the marginal implicit price of the licensed bear quota. As with the carnivore indexes, a *Z* test fail to reject the null hypothesis of pairwise equality between the licensed bear quota coefficients of the LAD and the OLS techniques. Hence, the marginal implicit price from LAD and OLS are not statistically different. LAD estimations shall be used as benchmark for policy purposes because least absolute deviations are resistant to data outliers, and therefore it yields narrower confidence intervals compared with OLS (Table 5).

while the marginal implicit price from LAD uses the median price (P=50), the marginal implicit price from OLS takes the average price (P=96.68). See the calculations in the Appendix for further details.

6. Conclusions

The hedonic pricing analysis conducted in this study pinpoints the effect of carnivore presence as a core constituent of hunting lease prices and quantifies the average impact on two specific societal agents: hunters, and landowners who lease out their properties for hunting activities. The wolf and lynx exert a negative and significant effect on hunting rental prices; nevertheless, the results for the wolf are not as robust as those for the lynx. Although the negative effect of the brown bear index is consistent across all estimations, the effect is not robust. The effect of the bear quota on hunting lease price is positive and significant across all estimations, indicating hunters' benefits from regulated brown bear harvesting. The higher impact of wolf on hunting lease prices seems reasonable given that the predation by wolf on ungulate game species is higher in biomass terms, and affects more game species (Wikenros et al. 2010, Andrén & Liberg 2015, Tallian et al. 2017). The weak evidence regarding the impact of brown bear abundance on hunting lease price is likely to be explained by the fact that the omnivorous brown bear mainly feed on vegetarian food sources and ants. Our results for lynx can be compared with Mensah et al. (2019) that used a completely different dataset on hunting leases, with 43 hunting plots in South Sweden. They obtain a marginal implicit price for an additional lynx buffer area of SEK 1.5 million (EUR 162 thousand), which is close to our results (SEK 1.31 million or EUR 141 thousand) from the median regression.

Based on our findings, the positive effect of brown bear license hunting on the price of hunting licenses for bear hunting seems to offset the negative effect of bear predation on ungulate game. This raises the question of whether increased license hunting for other species could have the same effect. If this would be the case, then there could be a positive net effect for hunters from an increase in carnivore populations, provided that the population increase was linked to a sufficiently large increase in license carnivore hunting. However, it is far from obvious that this would be the case. First, there is already license hunting for lynx, but no commercial market. One likely reason for the absence of a market is that people do not eat lynx meat and the pelt products are not allowed to sell (www.cites.org). Further, lynx is a relatively small animal, and compared to the brown bear is less seen as a hunting challenge. Thus, it has a comparatively low trophy value. In some cases, license hunting has been conducted on wolves, but it has not been associated with commercial sales of hunting opportunities. As the wolf is an iconic species with large symbolic value, this could substantially limit the possibilities to charge a price for the opportunity to hunt, whether the price is charged directly or included in the lease price. One can note that if license hunting of large carnivores is to be increased, there is strong support among hunters for ecologically well founded harvesting strategies, rather than strategies aiming a reduced human-carnivore conflicts, and a recognition that further training of hunting skills are necessary (Kaltenborn et al. 2013).

Further studies on the subject could broaden the analysis by using spatial-robust variance matrix estimates. Spatial correlation is a critical caveat of this study, and the omission of these spatial effects can lead to biased or imprecise estimates (Kim et al. 2003). Mensah & Elofsson (2017) show that disregard of spatial autocorrelation in lease prices may underestimate the value of ungulate game species by between 4% and 13%. Prospective research should thus further analyze the implications of spatial spillovers of large carnivores in the hunting context.

Further limitations of our study include the relatively small and single-year dataset, as well as the fact that we do not have information on all individual hunters that belong to the hunting team leasing a particular hunting ground. These factors prevent us from stepping into the second stage of the hedonic model in order to identify and estimate demand functions corresponding to each carnivore species. Therefore, the present study does not attempt to evaluate welfare changes in large carnivore populations and regulated carnivore hunting. Moreover, the absence of robust results for the bear index, and the relatively low robustness of the wolf index, imply that the calculations of marginal implicit prices are approximate values to be used cautiously for policy considerations. Also, our survey does not provide sufficient information to include a variable controlling for landscape characteristics of the hunting ground, such as altitude, share of open forest or the proportion of broadleaved trees, in the main regression. With clustering, we attempt to capture landscape heterogeneity by accounting for residual correlation within municipalities. However, the role of landscape characteristics of the hunting ground could be addressed in future research, especially considering the alleged importance of forest scenery to hunters, and the potential for landowners to influence this factor and, hence, the lease price.

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Appendix: Calculations of marginal implicit prices

Lynx and wolf territories:

For lynx and wolf we want to calculate marginal implicit prices related to an additional carnivore buffer. First, note that the carnivore index is computed as $C_k = \frac{n_k A_k}{M}$, where n_k is the number of carnivore buffers in a municipality for species k, A_k is the area of a buffer for the same species, and M is the area of the municipality. Hence, an additional carnivore buffer increases the index by $\frac{A_k}{M}$ units. Using Eq. (3) we then calculate the marginal implicit price as:

$$\frac{\partial P}{\partial C_k}\frac{\partial C_k}{\partial n_k} = \gamma_k P \frac{A_k}{\bar{M}}, \qquad (\mathrm{A.1})$$

where $\bar{M}$ is the average municipality area in our sample, see Table 2. For calculations using LAD estimates we use the median hunting lease price (P = 50), and for calculations using OLS estimates we use the average hunting lease price (P = 96.68), both can be found in Table 2. Equation (A.1) then yields the marginal implicit prices per hectare of hunting land. To obtain the total effect of an additional carnivore buffer, we need to sum over all huntable land in the municipality. We assume that the land suitable for hunting equals the sum of forest and agricultural land. These land types comprise 77% of the total Swedish territory (Statistics Sweden). The marginal implicit price of an additional carnivore buffer is then calculated by multiplication of the marginal implicit price per hectare and the area of huntable land in an average municipality, i.e. $\gamma_k P \frac{A_k}{\bar{M}} \bar{M} \cdot 0.77 = \gamma_k P A_k \cdot 0.77$.

Brown bear individual:

The brown bear index differs from the indexes for lynx and wolf. We first convert the index to numbers of individual brown bears. This is done by calculating the average number of brown bears per municipality, $\bar{n}_{BEAR}$, using the most recent estimation of the bear population in Sweden, which concludes there were 2 782 brown bears in 2013 (Swenson et al. 2017). The average municipality area in our sample (2 145 km^2) corresponds to 0.526% of the total size of Sweden (407 340 km^2, excluding lakes). Hence, there are on average 14.65 bears per municipality in the dataset. The average brown bear index, $\bar{C}_{BEAR}$, equals 0.12, see Table 2. Assuming a linear relationship between numbers and index, one additional bear in a municipality would imply an increase in the brown bear index equal to the ratio of the index and the average number, i.e., $\frac{\bar{C}_{BEAR}}{\bar{n}_{BEAR}} = 0.0084$ units. For the brown bear, the marginal implicit price is then calculated using Eq. (3) as:

$$\frac{\partial P}{\partial C_{BEAR}} \frac{\partial C_{BEAR}}{\partial n_{BEAR}} = \gamma_k P \frac{\bar{C}_{BEAR}}{\bar{n}_{BEAR}}. \qquad (A.2)$$

Similarly as for the two other species, we use the median hunting lease price for the LAD estimations and the average hunting lease price for the OLS estimation. Also similar to the above, we sum over all huntable land in an average municipality to obtain the total marginal implicit price of an additional bear individual.

Brown bear quota:

In the data section, we explained that the Q regressor is defined as the bear quota multiplied by the size of hunting area divided by the size of the county; hence:

$$Q = K \cdot \frac{W}{Z}, \qquad (A.3)$$

where K is the bear quota allocation in the county, W is the area of the hunting ground, and Z is the area of the county. We are interested in the effect of an additional quota allotment, i.e. a unit increase in K. We therefore insert (A.3) in Eq. (2), thereby obtaining (A.4), which we differentiate with respect to K:

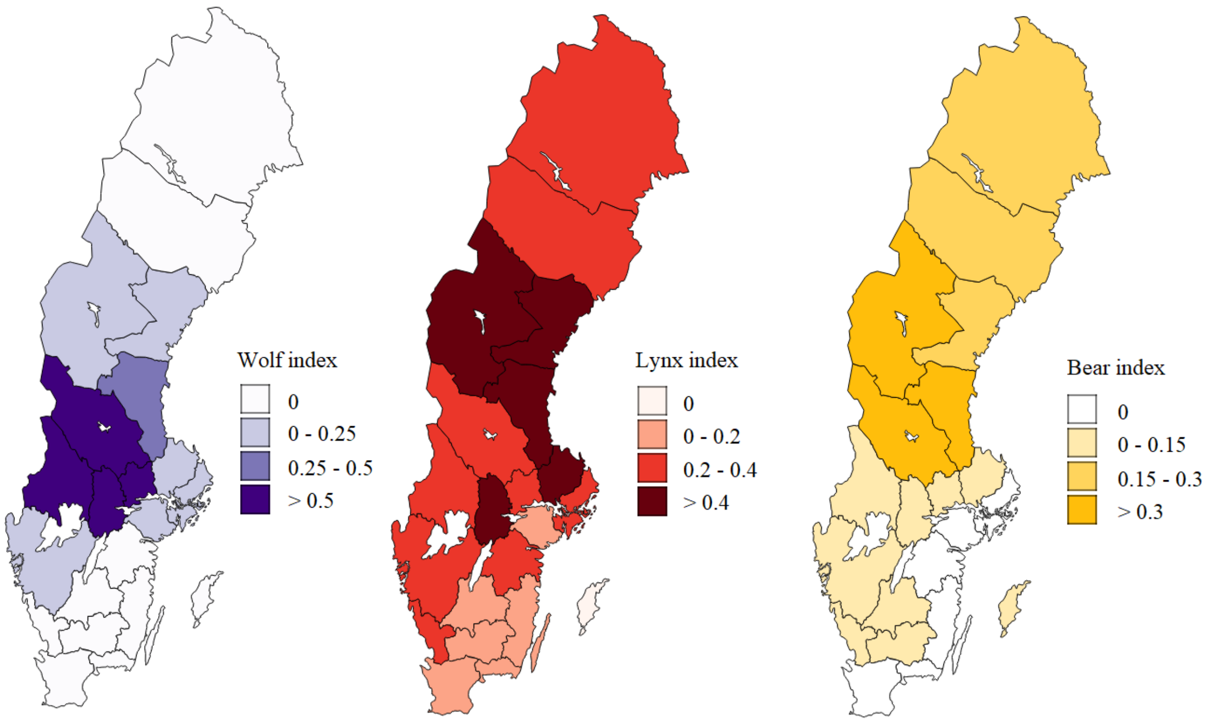
$$P = e^{\alpha+\beta \boldsymbol{H}+\gamma \boldsymbol{C}+\phi\left(K \cdot \frac{W}{Z}\right)+\delta \boldsymbol{A}+\theta \boldsymbol{G}+\varepsilon} \qquad (A.4)$$

$$\frac{\partial P}{\partial K} = \phi\left(\frac{W}{Z}\right)\left[e^{\alpha+\beta \boldsymbol{H}+\gamma \boldsymbol{C}+\phi\left(K \cdot \frac{W}{Z}\right)+\delta \boldsymbol{A}+\theta \boldsymbol{G}+\varepsilon}\right] = \phi\left(\frac{W}{Z}\right) P \qquad (A.5)$$

We hold W and Z constant at their averages, see Table 2, when computing $\frac{\partial P}{\partial K}$. Similarly as above, we use the median hunting lease price for the LAD estimations and the average hunting lease price for the OLS estimation, see Table 2. This yields the marginal implicit prices per hectare. In this case, however, an increase in the quota allotment in a county affects all hunting leases in that county. We therefore sum over all huntable land in the average county to obtain the marginal implicit price of an increase in the quota, which becomes $\phi\left(\frac{\bar{W}}{\bar{Z}}\right) P \bar{Z} \cdot 0.77 = \phi \bar{W} P \cdot 0.77$.

Appendix: Figures and tables

Figure A1. Carnivore densities on a county level (year 2014-2015).



Source: Carnivores densities are obtained from the Scandinavian database of large carnivore surveys (www.rovbase.no) for 2014/2015.

Figure A2. Map of municipalities where the hunting areas of the study are located.

Figures A3–A8. Densities of key variables with outlying observations. Figures on the right show the density of each variable after log transformation.

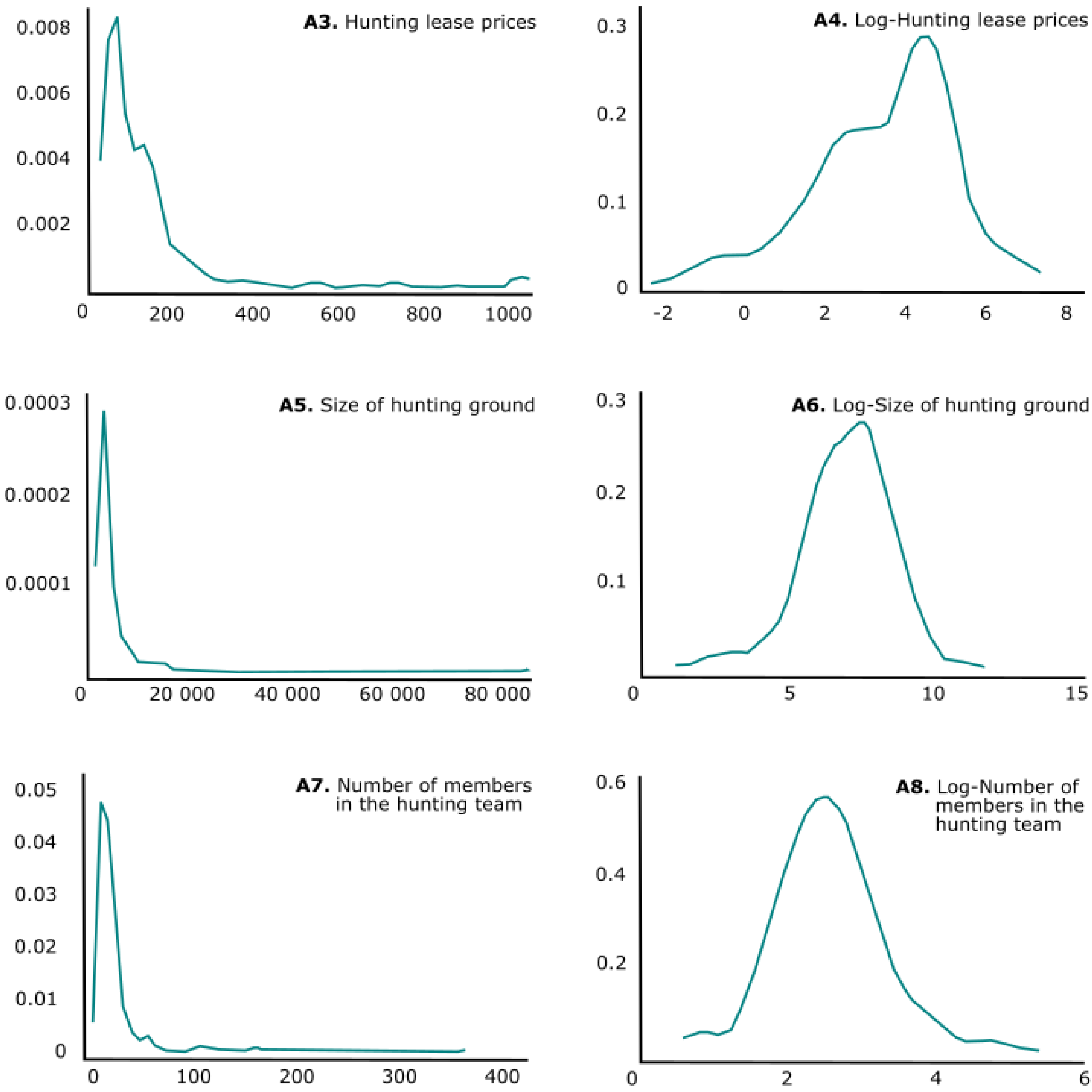


Table A1. Principal component analysis: Eigenvalues, correlation, and eigenvectors.

	Eigenvalues	Variance explained
Component 1	3.17	79.37%
Component 2	0.44	11.04%
Component 3	0.22	5.48%
Component 4	0.16	4.11%

	Component 1	Component 2	Component 3	Component 4
Forest_prod	-0.51	-0.42	0.21	0.72
Snow_depth	0.5	0.34	0.7	0.35

log(pop_dens)	-0.46	0.84	-0.18	0.21
log(distance)	0.52	0	-0.65	0.55

Table A2. OLS and LAD estimations with principal components.

Dependent variable: *log (price)* *Price: Hunting lease price per hectare for the hunting season 2014/2015)*	**OLS (1)**	**LAD (2)**	**LAD (3)**	**LAD (4)**
log(area)	- 0.34 *** (0.0709)	- 0.22 *** (0.0725)	- 0.20 *** (0.0581)	- 0.18 *** (0.0544)
log(members)	0.04 (0.1123)	0.06 (0.0787)	0.04 (0.0714)	
Component 1	- 0.31 *** (0.0718)	- 0.46 *** (0.0461)	- 0.48 *** (0.0396)	- 0.49 *** (0.0397)
Component 2	- 0.23 (0.1577)	- 0.35 ** (0.1404)	- 0.31 *** (0.0915)	- 0.31 *** (0.0854)
Wolf index	- 0.45 ** (0.2205)	- 0.32 (0.2347)	- 0.32 (0.2284)	- 0.31 (0.2277)
Lynx index	- 1.45 *** (0.5058)	- 1.35 *** (0.4046)	- 1.21 *** (0.3364)	- 1.27 *** (0.3687)
Bear index	- 0.72 * (0.4248)	- 0.36 (0.3728)	- 0.30 (0.2428)	- 0.26 (0.3671)
Bear_quota (adj)	2.28 *** (0.5832)	1.65 *** (0.2708)	1.61 *** (0.2547)	1.56 *** (0.2618)
log(income_pc)	- 0.04 (0.0859)	- 0.04 (0.0702)		
Company	- 0.23 (0.1921)	- 0.03 (0.1476)		
Intercept	6.26 *** (0.5595)	5.42 *** (0.5121)	5.18 *** (0.3477)	5.15 *** (0.3655)

R-squared (OLS): 0.4335
*Significance levels: *** 1%, ** 5%, * 10%.*
(Clustered standard errors)

Table A3. Variance inflation factor (VIF).

Variable	**VIF**
log(pop_density)	4.39

forest_prod	4.27
log(distance)	4.04
snow_depth	3.71
log(area)	1.94
log(income_pc)	1.89
bear	1.59
log(members)	1.39
bear_quota (adj)	1.22
company	1.17
wolf	1.13
lynx	1.11

Table A4. Matrix of correlation.

	log (area)	*log (members)*	*Forest prod*	*Snow depth*	*Wolf*	*Lynx*	*Bear*	*Bear quota (adj)*	*log (pop _dens)*	*log (income _pc)*	*log (distance)*
log(area)	1.00										
log(members)	0.42	1.00									
Forest_prod	-0.53	-0.24	1.00								
Snow_depth	0.47	0.08	-0.81	1.00							
Wolf	0.08	0.25	-0.01	-0.06	1.00						
Lynx	0.19	0.16	-0.14	0.08	0.11	1.00					
Bear	0.34	0.21	-0.46	0.39	0.08	0.14	1.00				
Bear_quota (adj)	0.39	0.20	-0.19	0.16	0.05	0.02	0.27	1.00			
log(pop_density)	-0.46	-0.11	0.61	-0.63	0.00	-0.23	-0.52	-0.19	1.00		
log(income_pc)	0.07	0.05	-0.13	0.12	0.11	0.12	0.12	0.03	-0.56	1.00	
log(distance)	0.52	0.16	-0.81	0.77	0.01	0.19	0.44	0.17	-0.71	0.21	1.00
Company	0.18	0.02	-0.25	0.25	0.12	0.07	0.20	0.11	-0.35	0.21	0.30