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The costs of livestock depredation by large carnivores

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Abstract

Livestock depredation by large carnivores entails economic damage to farmers in many parts of the world. The aim of this paper is to analyse and compare the costs of livestock depredation by carnivores across different carnivore species and regions. To this end, we estimate the government's compensation cost function. This study uses Swedish data on the county level over the period of 2001 to 2013. Compensation costs due to depredation by three large carnivores are considered: the brown bear (*Ursus arctos*), the wolf (*Canis lupus*) and the lynx (*Lynx lynx*). The results indicate that the costs of compensation for depredation by wolves, lynx and brown bears are determined by the densities of predators and livestock, the amount of forest pasture and the stock of preventive measures. There are considerable differences in marginal costs between predator species and counties, which have implications for policy.

Keywords: wildlife compensation, livestock depredation, lynx, wolf, brown bear, sheep.

The costs of livestock depredation by large carnivores

1. Introduction

Livestock depredation by large carnivores entails economic damage to farmers in different parts of the world due to lost, injured, and stressed livestock (Asheim and Mysterud, 2004; Baker et al., 2008; Howery and DeLiberto, 2004; Häggmark-Svensson et al., 2015; Laporte et al., 2010; Steele et al., 2013; Ramler et al., 2014; Sommers et al., 2010). The economic impact on the individual farmer varies substantially between different locations and predator species (Häggmark-Svensson et al., 2015). Large depredation costs may reduce peoples' tolerance for carnivores (Laporte et al., 2010), and hence jeopardize carnivore conservation efforts. Historically, conflicts between carnivores and human activities have led to low carnivore population levels in more densely populated regions. However, over the last century, this pattern was broken, as carnivores became increasingly protected by the law. Now, considerable populations are present in areas with high human density (Linnell et al., 2001). Several policies have been instituted to support carnivore conservation laws, typically aiming to reduce the economic risk to individual livestock holders, increase the tolerance towards carnivores, and reduce the incentives for illegal hunting (Nyhus et al., 2003). These policies include, for example, wildlife damage compensation to livestock holders, subsidies for prevention measures, and designated wildlife zones (Treves and Karanth, 2003).

Policy makers who wish to find a balance between the benefits and costs of carnivore preservation across regions and social groups need to understand the costs created by different predators and variations in these costs across different locations. However, most past studies do not systematically compare depredation costs for different predators or across space (Häggmark-Svensson et al., 2015). A few studies do make comparisons across space. Boman et al. (2003) analyse the optimal spatial distribution of wolf in Sweden using a constant marginal cost of wolf that differs between regions due to variations in hunting values, and Jones (2004) compares depredation costs across ten USDA farm production regions using input-output analysis. A couple of studies compare depredation costs for different predator species on an aggregated national level. Boman (1995) uses historical data on compensation costs, predator populations, and domesticated reindeer populations to estimate the social costs of politically targeted increases in large carnivore populations. Bostedt and Grahn (2008) add to the same topic, accounting for the presence of alternative, wild prey species when estimating the social cost.

The aim of this paper is to compare the costs of livestock depredation by carnivores across different carnivore species and regions. To this end, we estimate the government's compensation cost function, taking into account the roles that the populations of carnivores, livestock, and wild prey play in depredation. We recognize that not only the population numbers but also the physical opportunities for carnivores to get close to the livestock are relevant to assessing the magnitude of depredation and hence the compensation cost. We therefore account for the role of grazing on forest pastures with no fencing as well as the stock of specifically designed electrical fencing in the costs of depredation.

The study uses Swedish data on the county level over the years of 2001-2013. Compensation costs due to depredation by three large carnivores are considered: the brown bear (*Ursus arctos*), the wolf (*Canis lupus*) and the lynx (*Lynx lynx*). We limit the analysis to the costs of depredation for farm animals, thereby excluding damage to reindeer herders, for which the compensation system has a fundamentally different design (see, e.g., Zabel et al., 2011). Among farm animals, sheep are by far the most commonly attacked species (Elofsson et al., 2015) and therefore the focus of this paper. The costs of attacks by wolverines are excluded from the analysis because the wolverine primarily preys on reindeer, and very few attacks on sheep have been reported.

We use a mixed model approach with panel data to estimate a constant elasticity cost function. Possible differences in compensation level between counties are dealt with using Best Linear Unbiased Predictors, which give estimates of random effects for each county. The major contribution of the study is the consideration of landscape and county-specific factors, and the counteracting effect of preventive measures for the compensation paid. Further, we contribute by comparing the marginal costs of predators and livestock across regions, which allows us to contrast national cost effectiveness and farmers' perspectives. The results indicate that the costs of compensation for the depredation of wolves, lynx and brown bears are mainly determined by the densities of predators and livestock and, to lesser extent, the share of forest pasture and the stock of preventive measures. In contrast, the availability of alternative prey matters very little to the damage costs. There are considerable differences in marginal costs across predator species and counties. The results accentuate the challenges facing policy makers trying to reconcile different policy objectives, such as large carnivore conservation, pasture-based farming across the country, and national cost effectiveness.

The paper is organized as follows: In section 2, the Swedish wildlife damage compensation system is presented. In section 3, the theory and methodology are outlined. In section 4, the data are discussed. In section 5, the results are presented. In section 6, their implications are discussed. Finally, section 7 concludes the paper.

2. Swedish policies on carnivores, compensation and preventive measures

The current Swedish policy for wildlife damages was introduced in 1995. This policy makes clear that wildlife damage in general should be mainly prevented through hunting. However, large carnivores such as brown bears, lynx, wolves, wolverines, and golden eagles are protected by law (Environmental Protection Agency, 2006). Damage caused by these species on livestock carries the right to compensation if the livestock is part of a business. The compensation covers the costs for income foregone, provided that the economic loss can be verified by the livestock holder (Elofsson et al., 2015). The Wildlife Damage Center gives non-binding recommendations on compensation levels. These recommendations include a fixed compensation for sheep, varying between adults and lambs. Additional compensation can be achieved for organic farmers, farmers that participate in certain quality programs, or if the farmer claims that the lost animal is in gestation or otherwise particularly valuable. Further, it is recommended that cattle are compensated based on a case-by-case evaluation of the value. Veterinary costs are usually reimbursed against receipts. Other costs, such as those for additional work hours or lost environmental subsidies for natural grazing lands when predation impedes grazing, have successively become eligible for compensation, and the level of compensation for such factors is determined on a case-by-case basis. In each case, the final level of compensation to the individual farmer is decided by the county administration. Funds are allocated on an annual basis by the Environmental Protection Agency, EPA, to each county administration, based on their claimed needs for the purpose. In 2014, the total compensation for depredation amounted to 0.25 million EUR.

Concerns have been raised that wildlife damage compensation reduces the incentive to undertake proactive measures against wildlife damages, implying a risk for increased wildlife damages (Rollins and Briggs, 1996; Zabel et al., 2011). To counteract such effects, the Swedish policy includes subsidies for preventive measures in livestock farms. These subsidies are almost exclusively used for the installation of fences designed to keep carnivores out, so-called “carnivore electric fences”. In 2014, approximately 2 million EUR was paid as subsidies for preventive measures. The subsidies are jointly funded by the EPA and the Board of Agriculture. The funding from the Board of Agriculture has been available since 2010, when EU regulations made it possible to use the Rural Development Programs for this purpose, and is earmarked for counties with a stable wolf population (Wildlife Damage Center, 2013).

It is argued in the public debate that traditional forest pasture farming is threatened by increased carnivore populations in the area where this type of farming is practiced. Unfenced forest pasturing systems are particularly exposed to carnivore attacks, but they provide unique collective goods in terms of biodiversity, open landscapes and cultural value. The rules for wildlife compensation to forest pasture farms are the same as those for other livestock holders.

3. Methodology

3.1 Theory

Our aim is to investigate the government's short-run compensation cost function. The government can here be thought of as the Environmental Protection Agency (EPA), which is responsible for compensation funds.

The standard short-run cost function describes the relationship between costs, output levels, prices of variable inputs, and the levels of fixed and quasi-fixed inputs. The cost function approach is relevant as long as the decision maker can be assumed to minimize costs. Here, we assume that the EPA incurs costs for compensating farmers for wildlife depredation according to regulations that are fixed in the short run: regulations to protect large carnivores are decided at the parliamentary level for longer time periods (Government, 2013), and decisions on compensation payments in the individual case is, according to regulations in place, delegated to the county administrations. In addition, agricultural support schemes, such as the rural development program, which can be of importance for the number of livestock and hence the predation rate, can be seen as exogenous to the wildlife compensation scheme, and there are no clear links between the two policies. Consequently, even if the EPA wishes to minimize cost, it cannot do so in the short run, although this is evidently possible in the longer term. Instead, the EPA will have to pay a total compensation that is determined by a number of exogenous factors.

We take the output level to be the number of sheep, S, assuming that the ultimate aim of the compensation scheme is to maintain certain sheep farming activity. We assume that predators, P, prey on sheep. Elements from predator-prey models can be used to formulate the relationship between the number of predators and the number of sheep killed, K (Clark, 2010). Following Boman (1995) and Bostedt and Grahn (2008), we assume that the number of predators, as well as the availability of sheep, affects the number of sheep killed. Whereas Boman (1995) and Bostedt and

Grahn (2008) define the availability of livestock only as the number of livestock, we take into account that sheep availability is also determined by farming technology, z, including different farming regimes as well as the use of preventive measures.

Following Boman (1995), Bostedt and Grahn (2008), Zabel et al. (2014) and Skonhoft (2006), we assume that depredation has no feedback effects on the size of the predator populations. This assumption is motivated by the fact that sheep are not an important food source for the studied predators, and the number of sheep killed by all predators is only approximately 500 per year (Elofsson et al., 2015). Instead, the large carnivores' main food intake consists of wild prey, such as roe deer and moose, and semi-domesticated reindeer, in different compositions for different predators. For example, the brown bear is omnivorous and consumes meat from ungulates, reindeer is a relatively important food source (Persson et al., 2001; Karlsson et al., 2012). For the lynx, the most important prey is roe deer, but they also prey on reindeer where available and smaller mammals, such as hares (Odden et al., 2006; Liberg and Andrén, 2009). The main food source for the Swedish wolf population is moose, which constitutes approximately 95 per cent of the total biomass intake (Sand et al., 2008). We therefore assume that wild and semi-domesticated prey are a substitute for sheep and that if the population of wild prey and reindeer, m, increases, the depredation pressure on sheep falls.

In this context, it is also necessary to consider possible interaction effects between the different carnivore species. There is no indication that competition between carnivore species for livestock would be relevant for the compensation costs. Wikenros et al. (2010) find that the intensity of interference and competition for wild prey between wolves and lynx is low, and studies of the competition for prey between wolves and brown bears do not show any conclusive interaction effects with respect to wild prey species (Milleret, 2011). As mentioned above, only a small number of sheep are killed every year, which further reduces the likelihood of competition effects.

Following, e.g., Zabel et al. (2014), we assume that in the short run, the size of the sheep population is not affected by compensation payments. This is a simplification because if compensation exceeds (is below) farmers' actual cost increased (decreased) profits in the livestock sector provide incentives for larger (smaller) livestock holdings (Rollins and Briggs, 1996). Our assumption has support in the study of Berger (2006), who observed that wildlife policies have an insignificant impact on the sheep industry in the United States; instead, the development of the industry is determined by prices in the sector.

The Environmental Protection Agency's compensation cost, *C*, for wildlife depredation on sheep can then be expressed as follows:

$C = f\ [w, K]$, where
$K = g(S, P, m, z)$

and w is the unit compensation per sheep. Compensation costs are assumed to be increasing in S and P, and in the absence of predators, or sheep, the compensation cost is zero, i.e.C=f [w,K,0,P,z]=f [w, K,S,0,z]=0.

We do not have data on *w*, and even if we did, the inclusion of *w* in the analysis could give rise to substantial endogeneity problems in the econometric estimations. However, changes in rules and practices for compensation, and changes in the allocation of compensation funding across different counties, as described above, could matter for the total compensation cost. Such changes are indirectly accounted for in our model by the inclusion of dummies for each year.

3.2 Empirical model

Empirically, we estimate a constant elasticity cost function where *C* is the EPA's total compensation cost, *S* is the population/density of sheep, *y* is dummies for each year in the time series, *P* is the population/density of predator, *m* is the population/density of alternative prey species, and *z* is fencing practices[1]. The cost function is then defined by the following:
$C = \alpha S^{\beta} P^{\delta} m^{\varepsilon} z^{\theta}$.

In logarithmic form, the cost function can be expressed as follows:

$$log(C_{it}) = log(\alpha) + \beta \cdot log(S_{it}) + \gamma \cdot y_t + \delta \cdot log(P_{it}) + \varepsilon \cdot log(m_{it}) + \theta \cdot \log(z_{it}),$$

which is the function that we estimate econometrically using the natural logarithm. Here, the index *i* denotes the *i*=1,…,21 counties, and *t* denotes the year. Using the above logged cost function, we can interpret the estimated coefficient β as the output elasticity of the compensation cost, γ as the

[1] Alternative functional forms have also been investigated, and the results have shown that the logged cost function function performs better in terms of predictability.

elasticity of the dummies for each year, δ and ε as the elasticities of the compensation cost with respect to predator and alternative prey numbers, and θ as the elasticity of fencing practices.

In contrast to previous studies estimating cost functions for the large carnivores in Sweden, notably Boman (1995) and Bostedt and Grahn (2008), we do not treat Sweden on a national level as the unit of observation, with associated total populations of prey and predator species for each year. Instead, we make use of county-level data. This approach is advantageous because it implies a larger number of observations and allows for the analysis of how landscape and county-specific factors affect the compensation costs. To account for differences in the size of counties, we use the densities of both animal populations and costs, that is, we divide the population number of each species by the area of the county.

We use a time-series estimation technique called a mixed model. A mixed model includes both fixed effects and random effects. This model is useful when there are repeated observations of a subject over time, in this case, the counties (West et al., 2014). The standard errors are modelled as auto-regressive, meaning that the observation in a county is assumed to be correlated with the observation in the same county the year before. Given the limited number of observations it is not possible to investigate significant differences between counties with regard to the estimated elasticities. However, to account for possible differences between counties, we use so-called Best Linear Unbiased Predictors, which give estimates of random effects for each county (Robinson, 1991). The estimation technique is a restricted maximum likelihood, REML, and we run the estimations using the software SAS.

4. Data

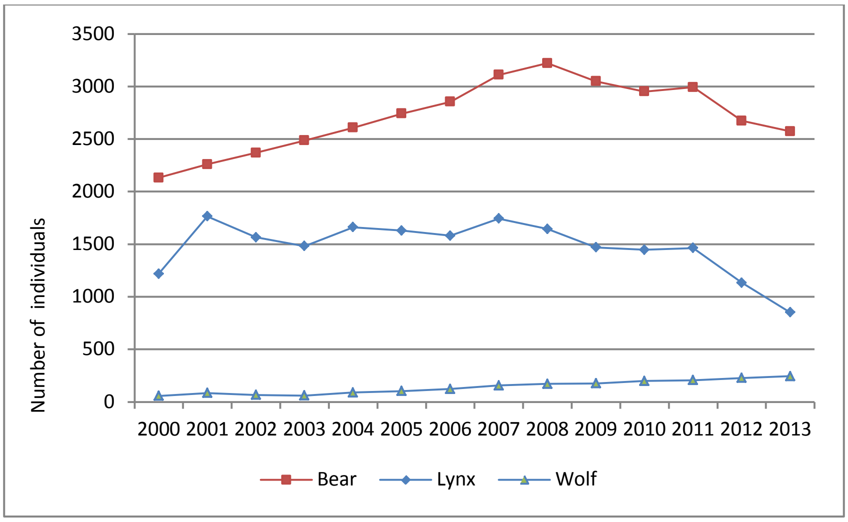
4.1 Data on animal populations

We use data on populations of carnivore species, wild prey species and domesticated and semi-domesticated animals in each county over time. The figures for lynx and wolves are obtained from the yearly inventories reported by the Swedish Wildlife Damage Centre and the Norwegian Hedmark University and are the best approximations available (Aronsson et al., 2003; Wabakken et al., 2004, Wabakken et al., 2005; Liberg and Andrén, 2006; Wabakken et al., 2006; Aronsson and Svensson, 2007; Aronsson et al., 2008; Wabakken et al., 2010; Andrén et al., 2010; Svensson et al., 2013; Zetterberg, 2014)[2]. Over the time period of interest, the brown bear populations in each county have been inventoried only in a single year, either 2001, 2004 or 2006, by the Scandinavian Brown Bear Research Project (Kindberg et al., 2009). The inventory data and population growth factors presented in Kindberg et al. (2009) are used to calculate the population number in all of the other years in the time series. Needless to say, there is a certain degree of uncertainty in all estimates of wildlife populations. In particular, the estimates for the brown bear population are uncertain mainly due to the difficulties in making inventories of traces in the wintertime because the brown bear goes into hibernation (Bostedt and Grahn, 2008; Kindberg et al., 2009).

Figure 1 shows the development of the total carnivore populations in Sweden for the years 2001-2013. The population of brown bears increased in the 2000s, but has declined slightly in recent years. All three carnivores are considered to have reached the minimum population levels for favourable conservation status as formulated by the government (Government, 2013). When population goals have been reached, licensed hunting has sometimes been allowed. The population of lynx reached its maximum at the end of the 2000s. After that, licensed hunting of lynx resulted in a reduction of the population. As the graph shows, the lynx population reached a bottom level in 2013, and for the winter 2014/2015, licensed hunting was not permitted. The lynx population then increased to approximately 1020 individuals in 2014. The number of wolves has increased over time; the population estimates used here summarized to approximately 270 individuals in 2013, which is exactly the minimum level for favourable conservation status as defined by the national government (Government, 2013).

[2]. These reports are henceforth referred to as *Wildlife Damage Center/Høgskolen i Hedmark.*

Figure 1. Animal population numbers 2001-2013



Sources: Wildlife Damage Center/Høgskolen i Hedmark; Kindberg et al. (2009)

Data on the population of sheep are obtained from the Board of Agriculture (2014). All farmers have to register their livestock, so these figures should accurately capture the actual population numbers. The total number of sheep in Sweden has increased from approximately 370,000 in 2001 to 507,000 in 2013. The increase seems mainly to be explained by the increased use of sheep for grazing on natural grazing land, where grazing is eligible for support from the Rural Development Program (Board of Agriculture, 2012).

Free-range forest pasturing at summer farms[3] implies a potentially higher risk of carnivore attacks. We do not have data on the number of sheep on forest pasture but use the share of grazing land classified as forest pasture in the EU farming support system, obtained from the Board of Agriculture (2015). The six counties where forest pasturing is practiced are all located in the northern and central parts of Sweden.

[3]. Refers to the term *fäbodbete* in Swedish.

Table 1. Average animal populations (number of individuals) and share of forest pasture per county 2001-2013

County	Brown bear pop.	Lynx pop.	Wolf pop.	Sheep pop.	Forest pasture (%)
Stockholm	-	20	1.2	18,356	-
Uppsala	-	77	0.2	17,639	-
Södermanland	-	21	-	22,118	-
Östergötland	-	17	0.3	39,600	-
Jönköping	-	9	-	24,139	-
Kronoberg	-	7	0.1	15,785	-
Kalmar	-	10	-	35,614	-
Blekinge	-	3	-	14,195	-
Skåne	-	2	-	49,687	-
Halland	-	5	-	20,562	-
Västra Götalands	-	75	9.3	71,836	-
Värmland	-	168	47.4	18,409	5.9
Örebro	-	91	17.8	17,327	-
Västmanland	-	66	5.3	10,312	-
Dalarna	279	126	38.1	14,588	51.5
Gävleborg	397	123	14.4	14,695	29.4
Västernorrland	190	87	1.7	9,103	12.3
Jämtland	820	223	3.4	9,822	65.3
Västerbotten	276	168	0.6	9,408	13.3
Norrbotten	756	178	1.0	8,327	-

Sources: Kindberg et al. (2009); Wildlife Damage Center/ Høgskolen i Hedmark, Board of Agriculture (2014): Board of Agriculture (2015).

The brown bear population is concentrated to the northern counties, whereas lynx are spread over the whole country, although appearing in smaller numbers in the south of Sweden. There are a few counties with comparatively large concentrations of wolves. This is related to the government policy of keeping the reindeer grazing areas in the north free of wolf, which implies that to reach the goal of a stable wolf population, other parts of the country must host larger numbers of wolf. The wolf is successively spreading to the southern counties, but there are so far no stable populations.[4]

With regard to alternative prey species, population data for reindeer are obtained from the Sami parliament statistics (Sami Parliament, 2015). The data are for the winter herd, whereas the summer herd is approximately 60 per cent larger. Reindeer are reported in only the three counties where the

[4]. It is not possible to take the logarithm of zero, but the observations with zero values are interesting, such as compensation in counties without a stable population of wolves in a given year. The rule of thumb adopted in this paper is to add 0.5 of the lowest value of the variable to all observations, which makes it possible to include the zero observations in the model.

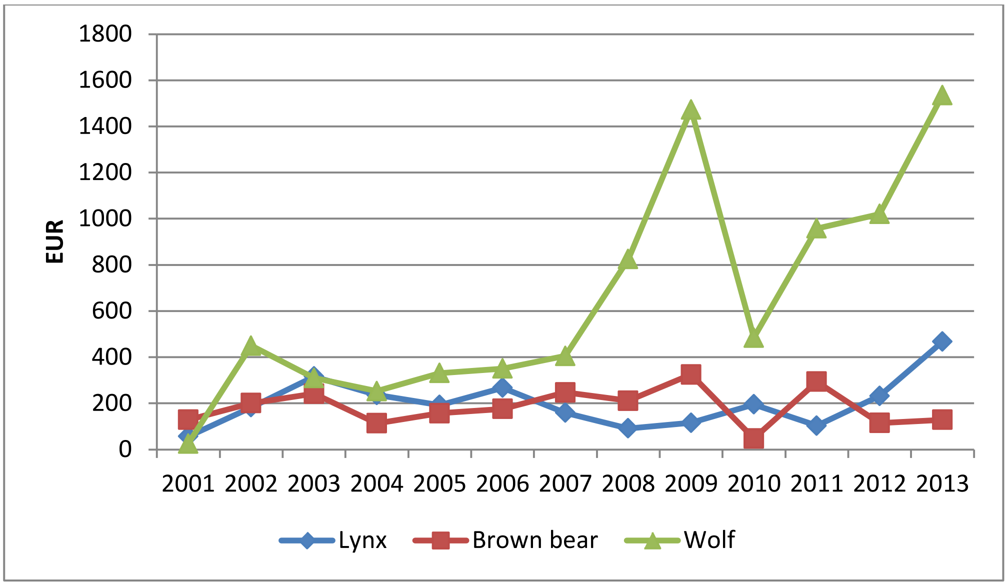
majority of Sami villages are located, although a few Sami villages are located in the neighbouring county of Dalarna. For roe deer and moose, there are no population data. Instead, we use the annual cull, which is a commonly used proxy for game species populations (see, e.g., Bostedt and Grahn, 2008). The cull of moose has to be reported to the county boards and is available online (County Administrative Boards, 2015). The cull of roe deer is reported annually on a voluntary basis by local hunter groups for their specific hunting grounds. The reported culls are used by the Swedish Association for Hunting Wildlife Management to produce an estimate of the bag rate in each county[5] (Elmhagen et al., 2011). These estimates are used as a proxy for the roe deer population. Summary statistics for all variables are presented in Table A1 in the Appendix.

4.2 Data on wildlife compensation and subsidies

The key variables on compensation for wildlife depredation of domestic animals and subsidies to preventive measures are obtained from the yearly reports by the Wildlife Damage Center for the years 2003-2013[6]. For 2001-2002, the database of wildlife damages of the EPA is used to complete the data (Environmental Protection Agency, 2015). All costs are deflated to 2013 prices and expressed in Euros.[7] Figure 2 shows the total compensation for depredations of each carnivore species in Sweden for each year in the time series.

[5]. The Swedish Association for Hunting and Wildlife Management, Wildlife Monitoring
[6]. Wildlife Damage Center, 2004; Wildlife Damage Center, 2005; Wildlife Damage Center, 2006; Wildlife Damage Center, 2007; Wildlife Damage Center, 2008; Wildlife Damage Center, , 2009; Wildlife Damage Center, 2010; Wildlife Damage Center, 2011; Wildlife Damage Center, 2012; Wildlife Damage Center, 2013; Wildlife Damage Center, 2014.
[7]. An exchange rate of 9.3798 SEK per Euro has been used, obtained from the Swedish Riksbank. http://www.riksbank.se/sv/Rantor-och-valutakurser/Manadsgenomsnitt-valutakurser/?y=2015&m=7&s=Comma [2015-08-30]

Figure 2. Total compensation for brown bears, lynx and wolves 2001-2013, in EUR



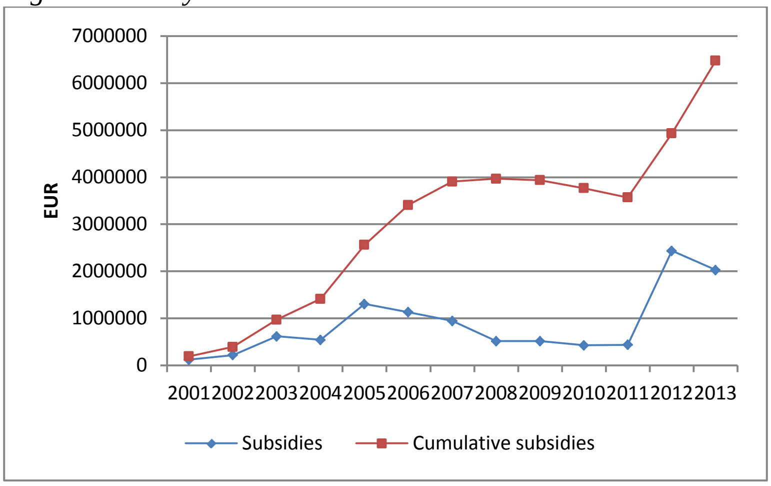
Source: Wildlife Damage Center/Environmental Protection Agency (2015).

The total compensations paid for lynx and wolf depredation are used as proxies for the compensation for sheep depredation, although they include compensation to all species over time, not just sheep. Wolves and lynx prey on some cattle and goats, approximately 10-20 in total for both species in each year of the time period, which can be compared to the 500 sheep killed per year. The proportion of different depredated livestock species is stable over time, and there are no clear differences between counties. The use of data that include costs for damages to animals other than sheep implies a tendency towards overestimating the costs of sheep depredation. In contrast, transaction costs for wildlife damages are not included in our data, such as costs associated with inspecting wildlife damages and administering compensation applications. The exclusion of these costs implies a tendency towards underestimation of the true cost.

For brown bears, the picture is different. They are omnivorous, which implies that they make use of several different food sources, such as berries and insects. The brown bear attacks a considerable number of beehives every year. Wildlife damages to beehives became eligible for compensation in 2008. The compensation to beekeepers is therefore deducted from the total compensation costs for damages caused by the brown bear.

There are governmental subsidies for carnivore electric fences, which are used to prevent carnivore attacks. Here, we use the stock of carnivore electric fences as a proxy for the use of preventive measures in a county. The stock of electric fences is calculated based on the amount of governmental subsidies for the purpose. To account for the deterioration of fences over time, we assume a depreciation period of ten years. Yearly and cumulative subsidies for the whole country are presented in Figure 3. The graph shows that the additional funding from the Board of Agriculture from 2010 led to a sharp increase in subsidies in the following years. For our analysis, we use the cumulative, depreciated subsidies with a one-year lag because subsidies are typically granted after the fence has been installed.

Figure 3. Yearly and cumulative subsidies 2001-2013



Source: Wildlife Damage Center/ Environmental Protection Agency (2015).

Even in counties with no stable population of wolves or bears, livestock depredation occurs by passing animals or animals without a marked territory. In Table 2, average compensation payments for each county and carnivore and average subsidies over the period of 2001-2013 are presented.

In the cost function for brown bears, only counties that have paid compensation in more than one year during the time period are included in the estimation. In the county of Norrbotten, no compensation for lynx depredation on livestock was paid during the period of interest. However, there is a large population of lynx in the county and it is therefore relevant to include in the analysis.

Part of the variation in granted compensation could be explained by compensation practices, which vary due to changes in the regulations and recommendations on what costs can be covered. The types of costs that are eligible for compensation have expanded over time. Since 2008, it has been possible to receive compensation for indirect costs, such as a loss of income from work other than farm work, as part of the wildlife damage compensation (Elofsson et al., 2015).

Furthermore, available funds from the wildlife damage grant allocated by the EPA to each county have varied over the years. Therefore, individual counties might sometimes have large funds available for compensation payments and sometimes have small funds. This can potentially affect compensation paid for a given amount of wildlife damage. To account for variation in compensation practices over time, we include year fixed effects in the model.

Table 2. Average compensation payments and average subsidies per county 2001-2013 in 2013 prices

County	Brown bear comp. (EUR)	Lynx comp. (EUR)	Wolf comp. (EUR)	Subsidies (EUR)
Stockholm	9	685	1,160	34,128
Uppsala	-	1,985	3,516	52,449
Södermanland	-	217	3,796	21,282
Östergötland	-	46	2,731	14,993
Jönköping	34	668	1,025	2,026
Kronoberg	-	131	157	6,383
Kalmar	-	146	2,364	3,941
Blekinge	-	135	1,148	1,011
Skåne	-	184	3,636	5,177
Halland	43	585	653	0
Västra Götalands	547	2,915	13,126	127,005
Värmland	249	4,658	8,687	117,374
Örebro	214	1,081	6,937	102,392
Västmanland	-	1,316	1,205	65,709
Dalarna	4,581	2,234	10,727	134,823
Gävleborg	4,452	1,072	2,611	115,658
Västernorrland	840	1,927	787	36,556
Jämtland	4,531	408	2,730	20,173
Västerbotten	1,692	701	413	2,844
Norrbotten	1,741	-	-	2,878

Source: Wildlife Damage Center/Environmental Protection Agency (2015).

5. Results

The results of the statistical analysis for all three carnivores, bears, lynx and wolves, are presented in Table 3. The results show that the costs of compensation for sheep are positively correlated to the densities of each carnivore and, for brown bear and wolves, to the sheep density.

For brown bears, the elasticity of sheep density indicates that a 1 per cent increase of the density of sheep results in an almost 1 per cent increase in the compensation cost. The share of forest pasture also has a significantly positive effect on compensation cost for brown bear depredation, whereas the compensation cost is negatively related to the lagged subsidies for preventive measures.

For lynx, the compensation cost is negatively related to the density of reindeer. The elasticity indicates that if the density of this alternative prey increases by 1 per cent, the cost of compensation decreases by approximately 0.2 per cent.

For wolves, the estimations indicate that if the sheep density increases by 1 per cent, the compensation cost increases by approximately 0.6 per cent.

The Best Linear Unbiased Predictors indicate that there are some differences between counties in the level of compensation. The Best Linear Unbiased Predictors (BLUPs) for the three cost functions are presented in Table A2 in the Appendix. The reference category for the year fixed effects was the year 2013. In the estimation for lynx and wolves, several years were significantly negative compared to 2013, as shown in Table A3 in the Appendix. For brown bears, the year fixed effects were little significant.

Table 3. Results of the compensation cost function estimations, including BLUPs, for brown bears lynx and wolves[8]*.*

	Brown bear compensation	Lynx compensation	Wolf compensation
Intercept	-4.35***	-4.33***	-3.01***
Brown bear density	1.056*** (0.277)		
Lynx density		0.389*** (0.108)	
Wolf density			0.479*** (0.092)
Sheep density	0.929* (0.468)	-0.277 (0.432)	0.603** (0.2243)
Moose and reindeer density[9]	-0.237 (0.407)		
Roe deer density		0.0506 (0.266)	
Reindeer density		-0.219*** (0.083)	
Moose density			-0.125 (0.247)
Share forest pasture	0.183** (0.087)	0.0450 (0.096)	0.0583 (0.091)
Lag subsidies	-0.243* (0.127)	-0.0351 (0.072)	0.0681 (0.089)

The marginal compensation costs for the different carnivore species, MC_P, can be calculated as follows:

$$MC_P = \left(\frac{\delta}{P}\right) * [\alpha + \beta * \ln(S_{it}) + \gamma * y_t +$$

$$\delta * \ln(P_{it}) + \varepsilon * \ln(m_{it}) + \theta * \ln(z_{it})]^{10}$$

[8]. *** indicates significance at the 1% level. ** indicates significance at the 5% level. * indicates significance at the 10% level. Standard errors are presented in parentheses. Aikaike's Information Criterion has been used to determine the model specifications presented here. Residual plots are presented in Figures A1-A3 in the Appendix

[9]. Reindeer populations are only reported in three counties. Even though the population measurements are differently constructed, we have chosen to simply aggregate them with the other prey species. When running different variations of the models, we obtain roughly the same estimates when including the prey variables separately and aggregated..

[10]. We rewrite the cost function on exponential form to simplify the calculation (note that z_1 is forest pasture and z_2 is subsidies and that z_2 is not part of the estimation for which MCs are calculated.) :

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In Table 4, the marginal compensation costs for brown bears, lynx and wolves are presented for each county, with Best Linear Unbiased Predictors included. The values of 2013 for each county are used in the calculations. The highest marginal cost for brown bears is found in counties with no stable population of brown bears, and relatively high densities of sheep, whereas marginal costs are low in counties with stable bear populations. The marginal cost is similar for counties with stable brown bear populations, but is slightly higher in the counties of Dalarna and Gävleborg, where forest pasturing is the most common and sheep densities are the highest among the counties with brown bears. The marginal cost of lynx varies between 1.8 and 87.5 EUR across the counties. The geographic distribution does not follow any clear pattern. In contrast, the marginal cost of wolves seems to be inversely related to the presence of wolves. Counties with relatively high densities of wolves have the lowest marginal cost, whereas marginal costs are particularly high in the southern counties with a low numbers of wolves but high densities of sheep. The highest estimate, 1,319 EUR, is observed in the southern county of Skåne, which also has the highest marginal cost for lynx.

Figure 4 shows the development of the marginal compensation costs of brown bears and lynx for different groups of counties over the time period of 2001-2013. For brown bears, the counties are divided into those with a stable population of brown bears and those with no stable population of brown bears. The graph shows that the marginal costs are higher in counties with no stable population of brown bears, which have higher densities of sheep. Marginal costs for both groups have increased over time.

Lynx are spread throughout the country but are more abundant in areas with a high roe deer density than in areas with a high reindeer density. In Figure 4, we distinguish between counties with roe deer (17 counties) and counties with reindeer (3 counties). Marginal costs for lynx have increased in the last few years and are approaching the levels of brown bears.

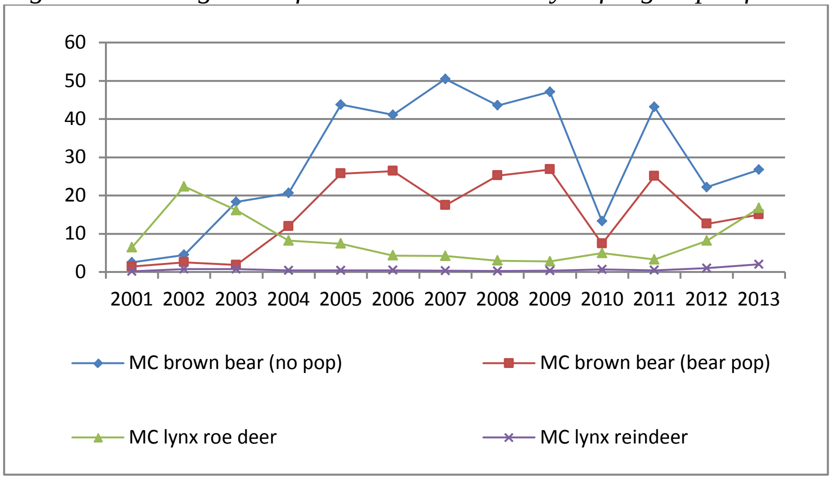
$$C_{it} = e^{(\alpha+\beta*\ln(s_{it})+\gamma*y_t+\delta*(\ln \mathrm{P_{it}})+\varepsilon*\ln(\mathrm{m_{it}})\mathrm{f}+\theta_1*\ln(z_{1it})+\theta_2*\ln(z_{2it-1}))}$$

Table 4 Marginal costs 2013

County	MC brown bear (EUR)	MC lynx (EUR)	MC wolf (EUR)
Stockholm		7.4	424
Uppsala		4.6	940
Södermanland		9.0	633
Östergötland		4.5	789
Jönköping		24.9	572
Kronoberg		22.5	423
Kalmar		8.0	862
Blekinge		4.7	519
Skåne		87.5	1319
Halland		11.3	675
Västra Götalands	170	33.9	243
Värmland	10.1	18.7	42
Örebro	26	3.6	75
Västmanland		7.4	59
Dalarna	18	29.7	81
Gävleborg	23	7.0	72
Västernorrland	6.2	11.7	92
Jämtland	9.5	2.1	80
Västerbotten	8.5	2.4	381
Norrbotten	4.1	1.8	200

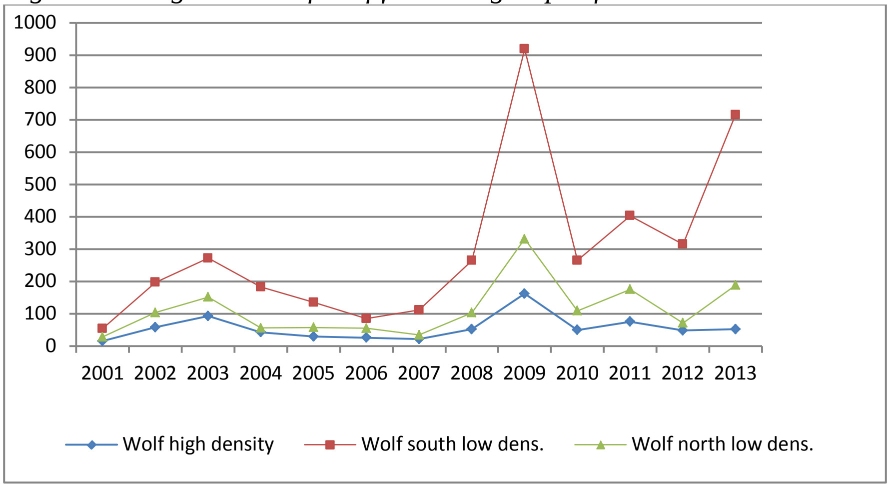
Source: Authors' calculations. Estimations include BLUPs.

Figure 4. Average MCs for brown bears and lynx for groups of counties 2001-2013



Source: Authors' calculations
Note: The marginal cost for a group of counties is calculated as the unweighted average marginal cost for the relevant counties.

Figure 5. Marginal cost of wolf for three groups of counties 2001-2013



Source: Authors' calculations
Note: The marginal cost for a group of counties is calculated as the unweighted average marginal cost for the relevant counties.

Figure 5 shows the development of the marginal cost for wolves for three groups of counties: first, counties in central Sweden with relatively high densities of wolf; second, counties in the south of Sweden with a high density of sheep but no regular occurrence of wolf; and third, counties in the north with low densities of both sheep and wolf. The graph indicates that the marginal cost of wolves have increased over the time period, particularly in the southern counties where it reached 700 EUR in 2013.

Finally, we calculate the marginal cost of sheep based on the wolf and brown bear cost functions where sheep density is significant. The results indicate that the total marginal cost per sheep is low, particularly in counties with few carnivores and high densities of sheep. The highest estimates are in the counties of Dalarna, Gävleborg and Jämtland where the sums of the marginal costs for sheep are 0.51-0.71 Euros. The marginal costs for sheep are presented in Table A5 in the Appendix.

6. Discussion

The results of the statistical analysis indicate that predator density and, to some extent, sheep density are important determinants for the compensation costs of large carnivores in Sweden. The density of wild prey is of little importance, given the current high abundance of wild prey. The compensation for lynx depredation is negatively affected by reindeer density but not by roe deer density. Reindeer herders and sheep farmers are both compensated for wildlife damages, and the net change in compensation to both groups that would result from an increase in reindeer populations depends on joint changes in depredation on both sheep and reindeer.

The results showed that the subsidies to preventive measures were only significant for the brown bear compensation cost. For brown bears, this result contrasts the results of earlier studies, where no impact of preventive measures on wildlife damages could be confirmed on the landscape scale (Treves et al., 2010; Wielgus and Peebles, 2014). For brown bears, the negative impact of preventive measures in combination with the positive impact of forest pasture suggests that the brown bear preys on sheep if they are easily accessible. The positive impact of forest pasture indicates a potential conflict between national aims to maintain both viable populations of large carnivores and traditional forest pasture-based livestock farming.

The significant year fixed effects confirm the presence of variations in compensation over time. Possible reasons are changes in rules, the implementation of rules, and the allocation of compensation funds. Only a few of the Best Linear Unbiased Predictors are significant. It is not possible to determine whether these differences between counties can be attributed to varying generosity by the county administrations, bottle-necks in the administration of compensations, or other factors that are not included in our estimations but promote or impede livestock depredation by large carnivores.

The marginal compensation costs vary between the different carnivore species. The marginal cost of lynx is relatively low all over the country. For brown bears and wolves, the marginal costs vary between different areas and are highest in counties with high densities of sheep but low densities of carnivores. The marginal cost of wolf reaches the highest levels of the three carnivores.

The management goals for wolves suggest that the population should be allowed to spread southwards. The high marginal costs of compensation for wolf depredation in the southern counties with no established wolf population but high densities of sheep implies that a larger number of carnivores in the southern parts of Sweden will lead to higher compensation costs. Therefore, the current concentration of carnivores to the central parts of Sweden, where the marginal cost of wolf is relatively low, seems rational from a national cost-effectiveness perspective. However, livestock holders in these areas are already bearing the highest costs. The results indicate that the marginal costs of sheep are highest in the counties with high densities of wolf. The highest estimate, 0.71 EUR in the county of Dalarna, represents only approximately 0.5 percent of the average price of a lamb (Agriwise, 2015). However, short-run profits in the sheep sector are close to zero (Agriwise, 2015), implying that this can be a substantial share of total profits. Thus, the concentration of carnivores in these areas can be unsuitable from an equity perspective.

The range of brown bears is expanding slower than those of lynx and wolves. The population increase is expected to occur in the same areas as now and expand to the county of Värmland. This expansion as well as single bears wandering to other counties implies relatively high marginal costs. The population of brown bear is approximately twice as large as the goal for favourable conservation status stated by the government. Our results suggest that an adjustment to the minimum level would imply decreased compensation costs for the EPA, but that savings would be small in the counties with stable populations.

The lower marginal costs for carnivores in counties with higher densities of carnivores could potentially be related to farmers' adoption of preventive measures. However, estimations could not confirm that subsidies for preventive measures were significant, except for brown bear depredation. This might be partly explained by the endogeneity of amounts paid in compensation and subsidies; funds for subsidies are mainly allocated to counties with high costs of compensation. In addition, subsidies are only a proxy for the actual occurrence of preventive measures. Farmers might apply other preventive measures, such as increased monitoring and taking in the animals at night. Further, even when electric fences are efficient at the farm level (Karlsson and Johansson, 2010), the effect on landscape level is not straightforward because if only few farmers have such fences, wildlife depredation might increase in neighbouring farms (Rollins and Briggs, 1996).

Due to the problems of inbreeding in the Swedish wolf population, an inflow of wolves from Finland and Russia could benefit the population in terms of viability (Government, 2013). Our results suggest that a consequential increase in the number of wolves in the reindeer herding areas could be associated with a considerable marginal cost in terms of depredation on sheep.

7. Conclusions

The wildlife damage compensation system aims to reduce economic damages to farmers by compensating for losses associated with livestock depredation. In this paper, we estimate a cost function for livestock depredation by large carnivores in Sweden. The analysis is conducted on a county level, which is the administrative decision level for wildlife damage compensation. Cost functions are formulated for the three most important large carnivores in regard to the depredation of sheep: brown bears, lynx and wolves. Data for the years 2001-2013 are used.

The results indicate that costs are positively related to the density of predators and to the density of sheep. For brown bears, costs are positively correlated with forest pasture and negatively related to sidies for preventive measures. The marginal costs of brown bear and lynx are relatively low, although there is variation across the country. The marginal costs for wolves also vary but are at a considerably higher level. In the southern counties with high densities of sheep, the marginal cost for wolves is estimated to be 700 EUR for 2013. The results accentuate the challenges facing policy makers trying to reconcile the policy objectives of conserving large carnivores and enabling pasture-based farming across the country while taking cost effectiveness and equity into account.

An interesting extension of the analysis would be to investigate the measurement of prevention efforts other than subsidies for carnivore electric fences and their impact on compensation costs. Further, it would be interesting to further explore the indications of differences between counties in compensation payments.

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Appendix

Table A1. Summary statistics

Variable	Average	Min	Max
Brown bear population number	138	0	998
Lynx population number	74.8	0	324
Wolf population number	7.34	0	96
Sheep population number	22,595	6,311	87,901
Roe deer population number	6,152	145	29,610
Moose population number	4,644	71	15,028
Reindeer population number	12,392	0	162,265
Share forest pasture (%)	9.28	0	72.0
County area (1000 Ha)	1,627	249	5,140
Compensation brown bear (EUR)	1,144	0	23,239
Compensation lynx (EUR)	1,068	0	17,697
Compensation wolf (EUR)	3,440	0	52,702
Subsidies to preventive measures (EUR)	43,174	0	604,277

Table A2. Best Linear Unbiased Predictor estimates

County	Brown bear	Lynx	Wolf
Stockholm		0.045	-0.082
Uppsala		0.197	0.368
Södermanland		-0.253	-0.098
Östergötland		-0.735*	-0.162
Jönköping		0.552	-0.081
Kronoberg		-0.377	-0.066
Kalmar		-0.350	0.116
Blekinge		-0.215	-0.025
Skåne		-0.004	-0.065
Halland		0.480	0.096
Västra Götaland	-0.121	0.713*	0.397
Värmland	-0.028	0.412	0.167
Örebro	0.151	-0.158	0.056
Västmanlanland		0.240	-0.474
Dalarna	0.318	-0.001	0.385
Gävleborg	-0.141	-0.441	-0.258
Västernorrland	-0.259	-0.124	-0.231
Jämtland	-0.019	0.109	-0.172
Västerbotten	0.032	0.082	0.102
Norrbotten	0.067	-0.175	0.028

Table A3. Year dummies

	Brown bear	Lynx	Wolf
2001	-2.07***	-2.02***	-1.90***
2002	-1.41*	-0.73	-0.64
2003	-0.03	-0.71	-0.48
2004	0	-1.04*	-0.93
2005	0.73	-1.09**	-1.30**
2006	0.63	-1.14**	-1.35**
2007	0.23	-1.17**	-1.57**
2008	0.53	-1.51***	-0.67
2009	0.62	-1.38**	0.55
2010	-0.72	-0.71	-0.74
2011	0.42	-1.08**	-0.34
2012	-0.27	-0.55	-0.78

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Table A4. Results of the compensation cost function estimations without BLUP

	Brown bear		Lynx		Wolf	
	Coefficient	Standard error	Coefficient	Standard error	Coefficient	Standard error
Intercept	-6.287***	1.156	-3.742***	0.932	-2.926***	0.884
Brown bear density	0.326***	0.079				
Lynx density			0.435***	0.105		
Wolf density					0.490***	0.086
Sheep density	0.921**	0.440	-0.446	0.389	0.601***	0.211
Moose and reindeer density[11]	0.126	0.330	0.231	0.254		
Roe deer density			-0.201*	0.064		
Reindeer density						
Moose density					-0.131	0.217
Share forest pasture	0.166**	0.071	0.048	0.074	0.057	0.079
Lag subsidies	-0.009	0.072	-0.047	0.069	0.057	0.085
2001	-1.116*	0.584	-2.090***	0.595	-1.941***	0.721
2002	-0.635	0.571	-0.833	0.593	-0.673	0.700
2003	0.388	0.547	-0.819	0.576	-0.504	0.672
2004	0.221	0.534	-1.129**	0.556	-0.944	0.656
2005	0.841	0.533	-1.168**	0.549	-1.315**	0.652
2006	0.655	0.529	-1.186**	0.542	-1.362**	0.649
2007	0.317	0.530	-1.246**	0.543	-1.588**	0.645
2008	0.567	0.529	-1.604***	0.546	-0.684	0.646
2009	0.848	0.529	-1.435***	0.542	0.535	0.646
2010	-0.589	0.525	-0.716	0.540	-0.750	0.641
2011	0.419	0.519	-1.102**	0.531	-0.355	0.638
2012	-0.200	0.471	-0.550	0.479	-0.790	0.599

[11]. Reindeer populations are reported only in three counties. Even though the population measurements are constructed differently, we have chosen to simply aggregate them with moose. When running different variations of the models where we include the prey variables one by one or in aggregates, they give roughly the same estimates.

Table A5. MCs for sheep 2013

	MC sheep bear (EUR)	MC sheep wolf (EUR)	Sum MC sheep (EUR)
Stockholm		0.012	0.012
Uppsala		0.026	0.026
Södermanland		0.015	0.015
Östergötland		0.01	0.01
Jönköping		0.012	0.012
Kronoberg		0.014	0.014
Kalmar		0.014	0.014
Blekinge		0.018	0.018
Skåne		0.013	0.013
Halland		0.015	0.015
Västra Götalands	0.001	0.029	0.03
Värmland	0	0.284	0.284
Örebro	0	0.138	0.138
Västmanland		0.065	0.065
Dalarna	0.344	0.368	0.712
Gävleborg	0.42	0.092	0.512
Västernorrland	0.131	0.06	0.191
Jämtland	0.497	0.076	0.573
Västerbotten	0.206	0.021	0.227
Norrbotten	0.332	0.017	0.349

Figure A1. Residual plots brown bear

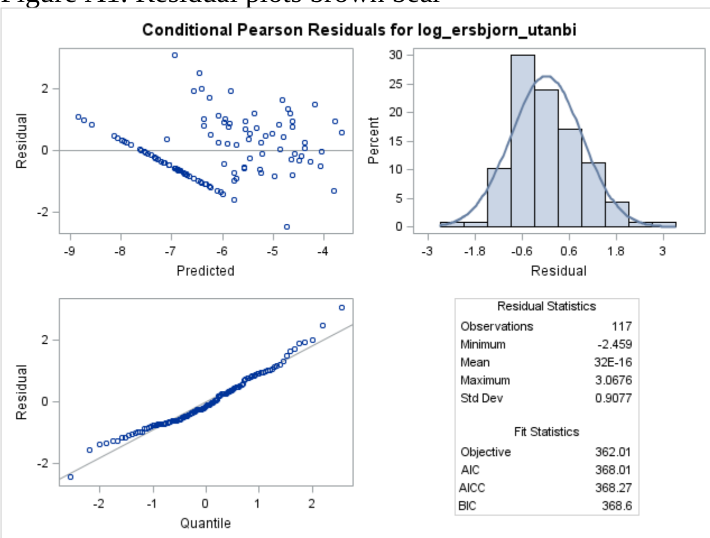

Figure A2. Residual plots lynx

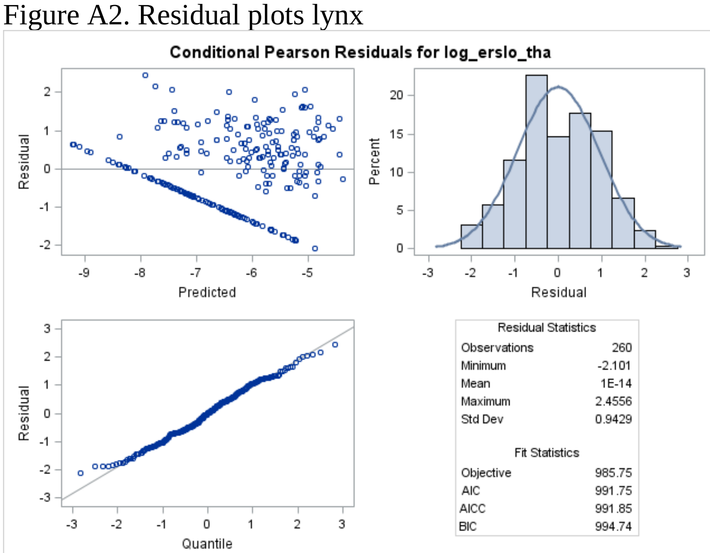

Figure A3. Residual plots wolf