Journal of Applied Ecology

Journal of Applied Ecology 2016 doi: 10.1111/1365-2664.12708

Patterns and correlates of claims for brown bear damage on a continental scale

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Summary

1. Wildlife damage to human property threatens human–wildlife coexistence. Conflicts arising from wildlife damage in intensively managed landscapes often undermine conservation efforts, making damage mitigation and compensation of special concern for wildlife conservation. However, the mechanisms underlying the occurrence of damage and claims at large scales are still poorly understood.

2. Here, we investigated the patterns of damage caused by brown bears *Ursus arctos* and its ecological and socio-economic correlates at a continental scale. We compiled information about compensation schemes across 26 countries in Europe in 2005–2012 and analysed the variation in the number of compensated claims in relation to (i) bear abundance, (ii) forest availability, (iii) human land use, (iv) management practices and (v) indicators of economic wealth.

3. Most European countries have *a posteriori* compensation schemes based on damage verification, which, in many cases, have operated for more than 30 years. On average, over 3200 claims of bear damage were compensated annually in Europe. The majority of claims were for damage to livestock (59%), distributed throughout the bear range, followed by damage to apiaries (21%) and agriculture (17%), mainly in Mediterranean and eastern European countries.

4. The mean number of compensated claims per bear and year ranged from 0·1 in Estonia to 8·5 in Norway. This variation was not only due to the differences in compensation schemes; damage claims were less numerous in areas with supplementary feeding and with a high

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proportion of agricultural land. However, observed variation in compensated damage was not related to bear abundance.

5. *Synthesis and applications*. Compensation schemes, management practices and human land use influence the number of claims for brown bear damage, while bear abundance does not. Policies that ignore this complexity and focus on a single factor, such as bear population size, may not be effective in reducing claims. To be effective, policies should be based on integrative schemes that prioritize damage prevention and make it a condition of payment of compensation that preventive measures are applied. Such integrative schemes should focus mitigation efforts in areas or populations where damage claims are more likely to occur. Similar studies using different species and continents might further improve our understanding of conflicts arising from wildlife damage.

Key-words: brown bear, damage compensation schemes, depredation, Europe, human land use, human–wildlife coexistence, human–wildlife conflicts, large carnivore conservation, supplementary feeding, wildlife management

Introduction

Coexistence of large carnivores and humans is a formidable challenge for conservationists world-wide (Treves & Karanth 2003). Carnivores cause economical and emotional losses due to, for instance, livestock depredation. They can be perceived as competitors for game and as a threat to human life, perceptions deeply anchored in human history and culture (Dickman 2010). At the same time, large carnivores are key species for ecosystem functioning and among the most admired animals (Ripple *et al.* 2014). This paradox often leads to deep societal conflicts between people that suffer losses and those aiming to conserve large predators (Young *et al.* 2010). Commonly, the mitigation of conflicts arising from damage to human property is addressed with compensation schemes to offset losses (Nyhus *et al.* 2005). In addition, measures to prevent damage, such as guarding animals or electric fences, are often subsidized to reduce losses (Baker *et al.* 2008; Rigg *et al.* 2011). Despite these efforts, the magnitude and economic impact of carnivore damage to human property is currently on the rise in many parts of the world (Treves & Karanth 2003; Can *et al.* 2014). Therefore, it seems crucial to improve understanding of the underlying mechanisms and factors associated with the occurrence of carnivore damage.

The association of damage incidence with ecological factors (Treves *et al.* 2011; Northrup, Stenhouse & Boyce 2012), as well as population management and demographic aspects (Kavčič *et al.* 2013; Wielgus & Peebles 2014), has received increasing attention. However, most studies have focused on the local or regional scale, while few have followed a more integrative approach across populations and different management scenarios (Kaczensky 1999; Berger 2006; Can *et al.* 2014). Many large carnivore populations are transboundary, and conflict management usually varies among countries due to, for example, differences in conservation status, public attitudes or livestock husbandry practices (Kaczensky 1999; Swenson & Andrén 2005). Therefore, comparative analyses at a broad scale are essential for disentangling the socio-economic and environmental factors related to damage occurrence in order to achieve effective conservation policies.

The study of conflicts generated by a generalist species such as the brown bear (*Ursus arctos*) is particularly interesting. After centuries of persecution and decline, most populations in Europe have experienced recent recovery and the brown bear is currently the continent's most abundant large carnivore (Chapron *et al.* 2014). The brown bear inhabits a wide range of habitats and its broad diet often includes anthropogenic food, such as livestock, crops and beehives (Bojarska & Selva 2012; Can *et al.* 2014).

Landscape features, such as forest composition, influence bear occurrence (Naves *et al.* 2003; Fernández *et al.* 2012), as well as the availability of natural foods, which is known to affect damage incidence in several bear species (Gunther *et al.* 2004; Garshelis & Noyce 2008). Bear damage is necessarily associated with human activities; for instance, the presence of agricultural lands and high human densities are related to a higher occurrence of bear damage claims (Wilson *et al.* 2006; Northrup, Stenhouse & Boyce 2012). At small scales, the number of claims has sometimes been found to be positively related to the number of bears (Garshelis & Noyce 2008; Mabille *et al.* 2015), and some countries have established culling quotas in order to keep a 'tolerable' number of bears (e.g. Huber *et al.* 2008b). Supplementary feeding may divert bears from preying on livestock, but can also promote nuisance behaviour, which increases the level of conflict (Gray, Vaughan & McMullin 2004). Reintroduced populations expand into areas where bears were extirpated and where traditional prevention practices no longer exist, leading to high damage incidence (Stahl *et al.* 2001). Finally, we expect that wealthier countries and regions could more easily afford the costs of compensating damage claims and, therefore, that the economic activity in regions where bears exist would have a positive effect on the number of compensations.

In this study, we aim to improve knowledge of human–bear interactions across different scenarios at a continental scale. As the first step, we characterized the compensation schemes in Europe, since they are pivotal to the number of claims (e.g. Swenson & Andrén 2005). Secondly, we compiled brown bear damage claims across Europe in 2005–2012 to characterize the patterns of compensated claims across bear populations. Finally, we explored the factors associated with damage claims across those countries and regions that use similar compensation schemes. Specifically, we evaluated status and management aspects of the bear populations, landscape features, such as forest availability and human land use, and socio-economic factors.

Materials and methods

BROWN BEAR POPULATIONS AND MANAGEMENT UNITS

At the time of the study, the distribution of the brown bear in Europe was clustered in 10 populations spanning 26 countries (Fig. 1, Table 1 and Table S1, Supporting information). Population sizes ranged from <50 bears in small isolated populations, such as the Pyrenean or Apennine, to several thousand individuals in larger ones, such as the Carpathian and Scandinavian populations (Chapron *et al.* 2014). Except for the Apennine and Cantabrian populations, all were transboundary, that is spanning more than one country. Some countries, such as Greece and Italy, held more than one population (Fig. 1). Actions related to the monitoring and management goals of brown bear populations, such as compensation payments, differ between countries and regions. Thus, we defined our study areas as management units (*sensu* Linnell, Salvatori & Boitani 2008), based on the distribution of each bear population or subpopulation overlaying national, regional or county borders (Fig. 1).

COMPENSATED CLAIMS FOR BEAR DAMAGE

We searched for information on the types of compensation schemes and data on compensated claims (claims hereafter) for damage caused by brown bears between 2005 and 2012 across Europe. We obtained data from national and regional wildlife agencies and published literature and reports, as well as from researchers and practitioners. The collected data contained information on the location, year, type of damage and the number of items damaged, that is the number of killed animals, destroyed beehives, fruit trees and silages, and hectares or tons of crop damaged. Damage claims were assigned to one of the following categories: (i) damage to livestock, including sheep, goats, cattle, reindeer, pigs, horses and donkeys; (ii) damage to apiaries, including beehives and bee colonies; (iii) damage to agriculture,

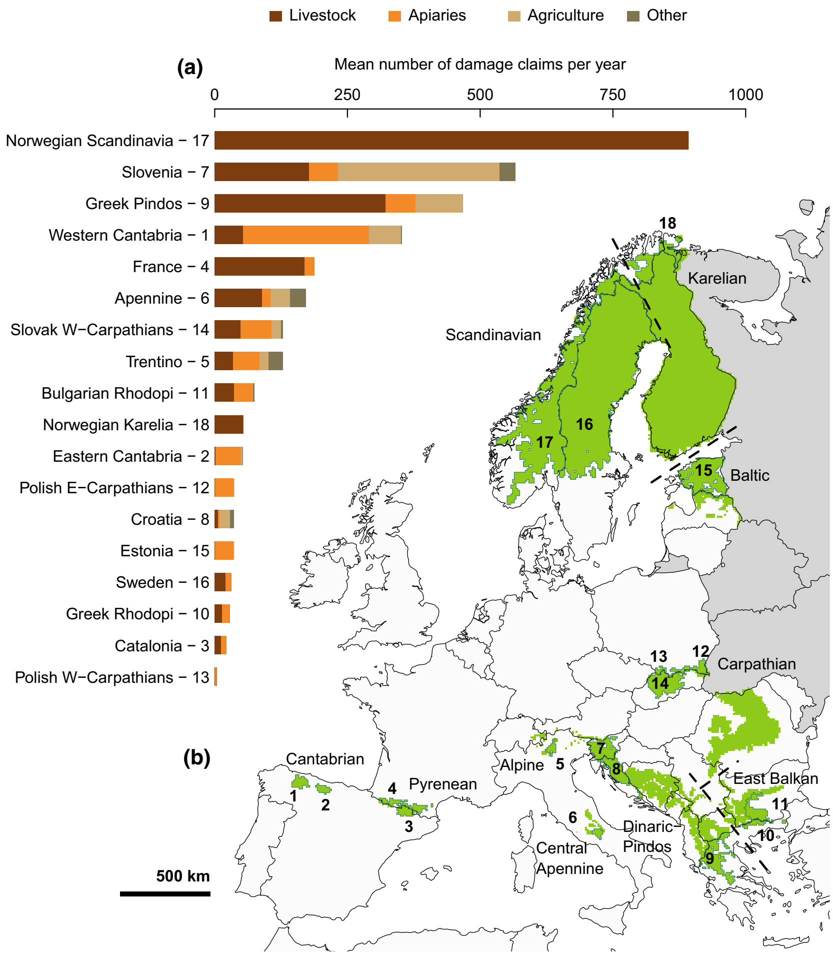

Fig. 1. (a) Average number of damage claims per year in 18 European management units in 2005–2012 and (b) distributions of European brown bear populations (from Chapron *et al.* 2014) and the management units included in this study. Blue lines in (b) delimit the studied management units. Countries with grey colour had no bear distribution data.

Table 1. Characterization of the studied brown bear populations in European countries in the period 2005–2012. Population estimates and trends, as well as information on the compensation systems and bear management, are provided. References are listed in Table S1

Population	Country	Bear population size	Trend	Compensation system	Years compensating damage	Bear harvesting	Supplementary feeding	Bear reintroduction
Cantabrian	Spain	223 (183–279)	+	Public administration at regional level	>30	No	No	No
Pyrenean	Spain	25 (shared with France)	+	Public administration at regional level	>30	No	No	Yes
	France	25 (shared with Spain)	+	Public administration at regional level	>30	No	No	Yes
Central Apennine	Italy	51 (47–66)	0	Public administration at regional and local levels	>30	No	No	No
Alpine	Italy	33	+	Public administration at regional and national levels	>40	No	No	Yes
	Switzerland	0–2*	0	Public administration at national and regional levels	≥10	No	No data	No
	Austria	5	–	Public administration and hunter associations at regional levels	20–30	No	Yes	No
	Slovenia	396–480†	+	Public administration at national level	>50	Yes	Yes	No
Dinaric Pindos	Slovenia	396–480†	+	Public administration at national level	>50	Yes	Yes	No
	Croatia	1000	+	Public administration at national level and hunter associations at local level	<15	Yes	Yes	No
	Bosnia & Herzegovina	550	+	No data	No data	Yes	Yes	No
	Serbia	70–80	+	Public administration at national level and local levels	No data	No	Yes	No
	Montenegro	270	+	No data	No data	No data	No data	No
	Albania	180–200	+	None	No data	No	No data	No
	Macedonia	160–200	+	No data	No data	No	No data	No
	Greece	350–400	+	Semi-public administration at national level	20–30	No	No	No
Eastern Balkans	Greece	30–40	+	Semi-public administration at national level	20–30	No	No	No
	Bulgaria	530–590	+	Public administration at national level	≥10	Yes	Yes	No
	Serbia	6–10‡	0	Public administration at national and local levels	No data	No	Yes	No
Carpathian	Serbia	6–10‡	0	Public administration at national and local levels	No data	No	Yes	No
	Romania	6000	0	Public administration and hunter associations	≥20	Yes	Yes	No
	Ukraine	300–400	0	None	None	No	Yes	No
	Poland	95	0	Public administration at regional level	≥15	No	Yes	No
	Slovakia	800	0	Public administration at regional level	>50	Yes	Yes	No
	Czech Republic	2–5*	0	Public administration at regional level	No data	No	Yes	No
	Hungary	0–2*	0	Public administration	≥15	No	Yes	No
Baltic	Belarus	60–100	0	None	None	No	Yes	No
	Latvia	10–15	0	None	Only in 2007	No	Yes	No
	Estonia	600–700	+	Public administration at national level	<10	Yes	Yes	No
Karelian	Finland	1600–1800	+	Public administration	20–30	Yes	Yes	No
	Norway	46	+	Public administration at national level	20–30	Yes	No	No
Scandinavian	Norway	105	+	Public administration at national level	20–30	Yes	No	No
	Sweden	3300	+	Public administration at national level	≤20	Yes	No	No

*Occasional presence.
†The number of individual estimated for the whole Slovenian territory in both populations.
‡The number of individuals estimated in both Serbian populations.

such as to fruit trees, silages, crops and other agricultural products; and (iv) other kinds of damage, ranging from backyard poultry and rabbits to fish ponds and construction materials, such as windows or fences (Tables S2–S4).

For those management units with similar compensation schemes, and to allow for comparisons, we calculated the damage-to-bear ratio (*damage ratio*), defined as the number of claims averaged across six years (within the period 2005–2012), and divided by the estimated number of bears in the respective management unit (Table 2). We used the average values for that period to reduce the effects of fluctuations and trends in the number of claims and bears (e.g. Garshelis & Noyce 2008; Bautista *et al.* 2015). Estimations of the number of bears for each management unit were extracted from the literature (Table S1). The damage ratio indicates the mean number of claims compensated per bear and year in each management unit and was calculated also for each of the four damage categories described above. We also quantified the mean number (± 1 SD) of sheep and beehives lost per claim for each year and then averaged for the study period to compare the severity of single damage claims among management units (Table S5).

CORRELATES OF DAMAGE CLAIMS

To test the association between bear damage claims and different ecological and socio-economic variables, we formulated five non-exclusive hypotheses including a total of 10 variables (Table 3). We created a 5 × 5 km grid and delimited the previously selected management units based on bear distributions from Chapron *et al.* (2014). We considered occupied bear range to include areas of permanent as well as occasional presence, as damage occurs in both, and we calculated accordingly the area of each management unit (Table S6). The explanatory variables tested under the bear population size and the management hypotheses were extracted from the literature and corroborated by collaborators (Tables 1–3, Tables S1 and S6).

The forest availability hypothesis included the forest cover (%) and the length of forest ecotones with shrubs and pastures (metres per hectare) as explanatory variables of the number of claims, while the human land-use hypothesis included agricultural cover (%) and human population density (inhabitants per km^2, Table 3). We estimated the value of each of these variables in each 5 × 5 km cell and then calculated the average for each management unit. Forest and agricultural cover and the length of forest ecotones were derived from the Corine Land Cover digital map for Europe (100 m resolution; CLC2000) available at http://www.eea.europa.eu/data-and-maps/data/corine-land-cover-2000-raster-3, and human population density was derived from the gridded world population data set (CIESIN 2005).

To test the economic hypothesis, for each management unit we calculated the gross domestic product (GDP) expressed in purchase power standard (PPS) per inhabitant and the GDP in millions of PPS per km^2. The former (GDP.PPS per inhabitant) is an indicator of the economic wealth and the latter (GDP.PPS per km^2) of the economic activity relative to the area of a given region. GDP at current market price expressed in PPS per inhabitant and in millions of PPS was extracted from the Eurostat data set (http://ec.europa.eu/eurostat/data/database). Eurostat provides both economic indicators at three nested territorial units called NUTS (Nomenclature of Territorial Units for Statistics), which are comparable among European regions. We extracted the variables at the finest territorial resolution available: NUTS-3. To calculate GDP.PPS per km^2, we divided GDP in millions of PPS of each NUTS-3 by its area (km^2). Finally, we averaged both economic indicators across the set of NUTS-3 that covered each management unit.

To test the forest availability, human land-use, management and economic hypotheses, we used generalized linear mixed-effect models (GLMMs) taking the number of claims per bear in each

Table 2. Brown bear damage ratios in selected management units in Europe, estimated as the mean number of damage claims (± 1 SD) compensated per bear and year in different periods between 2005 and 2012. For the Greek Pindos, Greek Rhodope and Bulgarian Rhodope, we used a 4-year period according to the changes in the compensation schemes (Karamanlidis *et al.* 2011; A. Dutsov 2014, unpublished data). The estimated number of bears for each unit used in the calculations is given in the Table S6

		Damage ratio (mean ± SD)				
Management units	Years	Total	Livestock	Apiaries	Agriculture	Other
Western Cantabria	2005–2010	1·7 ± 0·47	0·26 ± 0·045	1·2 ± 0·37	0·30 ± 0·15	0·0057 ± 0·014
Eastern Cantabria	2005–2010	2·8 ± 1·1	0·070 ± 0·043	2·6 ± 1·1	0·16 ± 0·082	0·0088 ± 0·021
Catalonia*	2005–2010	0·87 ± 0·25	0·47 ± 0·23	0·40 ± 0·31	0	0
France	2005–2010	7·5 ± 2·2	6·8 ± 1·8	0·72 ± 0·42	0	0
Trentino*	2005–2010	4·4 ± 1·8	1·2 ± 0·38	1·7 ± 0·96	0·57 ± 0·23	0·96 ± 0·52
Apennine	2005–2009	3·4 ± 1·4	1·8 ± 0·62	0·31 ± 0·27	0·73 ± 0·43	0·58 ± 0·36
Slovenia	2005–2010	1·2 ± 0·37	0·39 ± 0·098	0·12 ± 0·053	0·68 ± 0·28	0·066 ± 0·028
Greek Pindos	2007–2010	1·3 ± 0·13	0·86 ± 0·10	0·15 ± 0·026	0·24 ± 0·076	0
Greek Rhodope	2007–2010	0·82 ± 0·36	0·41 ± 0·24	0·41 ± 0·28	0	0
Bulgarian Rhodope	2009–2012	0·24 ± 0·12	0·12 ± 0·027	0·11 ± 0·11	0·0056 ± 0·0054	0·0063 ± 0·0045
Polish West Carpathians	2005–2010	0·11 ± 0·076	0·029 ± 0·037	0·074 ± 0·055	0	0·0049 ± 0·012
Polish East Carpathians	2005–2010	0·60 ± 0·63	0·019 ± 0·024	0·58 ± 0·63	0	0
Slovak West Carpathians	2007–2012	0·16 ± 0·054	0·062 ± 0·0093	0·072 ± 0·032	0·023 ± 0·016	0·0042 ± 0·0019
Estonia	2007–2012	0·053 ± 0·013	0·0015 ± 0·0024	0·042 ± 0·024	0·00075 ± 0·0013	0
Norwegian Scandinavia	2005–2010	–	8·5 ± 1·3	–	–	–
Norwegian Karelia	2005–2010	–	1·2 ± 0·63	–	–	–

'–' indicates no data are available.
*Corresponding bear population in Table S2.

Table 3. Hypothesized effects on the number of brown bear damage claims in Europe and explanatory variables tested in each hypothesis

Hypotheses	Description	Explanatory variables	Predicted effect
Bear population size	The number of bears affects the number of damage claims	The number of bears (transformed to its natural logarithm)	+
Forest availability	Forest and their ecotones are suitable bear habitats that provide natural foods; the number of claims is affected by food availability	Forest cover (%)	−
		Length of forest ecotones with shrubs and pastures (m ha^{-1})	−
Human land use	Claims for damage happened where human activities and bears meet	Agricultural cover (%)	+
		Human density (inhabitants per km^2)	+
Management	The management practices affect bear behaviour and can influence how prone bears are to cause damage to human properties	Reintroduction (yes/no)	+
		Supplementary feeding (yes/no)	+
		Harvesting (yes/no)	−
Economic	The wealth of the regions influences the eagerness of rural stakeholders to claim damages since the costs of compensations are easily covered	Gross domestic product expressed in power purchase standard per inhabitant (GDP per inhabitant)	+
		Gross domestic product expressed in million of power purchase standard per km^2 (GDP per km^2)	+

year as the dependent variable (see the periods in Table S6). We fitted every model using a negative binomial error distribution and included the management unit as a random factor. We performed GLMMs separately for each of the following response variables: the total number of damage claims, claims for livestock damage and claims for damage to apiaries. We excluded damage to agriculture from the analyses due to the low number of cases. We set the number of bears and the surface of the management units (km^2) as offsets to account for differences in the size and distribution area of bear populations. We first transformed both variables to their natural logarithms and included their sum as the offset term in the model formula.

We used an information-theoretic approach for model selection to rank hypotheses (Anderson, Burnham & Thompson 2000). Specifically, we examined a set of *a priori* specified models, based on the hypothesized effects of the explanatory variables (Table 3). Each hypothesis was tested running a full model (all explanatory variables), as well as nested univariate models for each variable. In order to reduce the problems associated with collinearity, we did not include variables highly correlated ($r > 0{\cdot}7$) within each hypothesis. We limited the number of variables per model to a maximum of three to avoid overfitting, and limited the number of models tested to reduce the risk of finding spurious correlations. We ranked the resulting set of candidate models according to the small sample-unbiased Akaike Information Criterion (AIC_c). To assess the importance of each hypothesis, we calculated the 'hypothesis weight' as the sum of the AIC_c weights of the subset of models composing each hypothesis.

For the bear population size hypothesis, we tested whether the estimated number of bears explained the observed variation in the number of claims in each year across management units. We also used GLMMs with the management unit as a random factor, a negative binomial error distribution and the same response variables. The natural logarithm of the area of the management unit (km^2) was included as an offset in these models.

We standardized the explanatory variables to zero mean and unit variance to allow for the comparison of effect sizes between variables. All statistical analyses were performed in R (version 3.1.2, R Development Core Team 2014) using the package glmmADMB for fitting GLMMs (Fournier *et al.* 2012) and the package MUMIN for model selection (Barton 2015).

Results

BROWN BEAR MANAGEMENT AND COMPENSATION SCHEMES

Brown bear management is highly heterogeneous across the 26 European countries where the species occurs. For example, while bears are autochthonous and legally hunted in Croatia and Estonia, they have been reintroduced under full protection in Trentino (Italy) and the Pyrenees (France and Spain, Table 1). Most European countries covered in this study have a compensation system for brown bear damage, with the exception of Latvia, Belarus, Ukraine and Albania. Compensation is established by law and, in most cases, managed by the public administration at national or regional levels (Table 1).

Compensation in Europe is mostly paid *a posteriori* based on expert-verified losses. Typically, the affected person is obliged to declare alleged bear damage to the competent authority within a defined time limit. The authority then sends qualified staff to assess the cause of damage and its costs, and to complete a technical report. Based on this report, the competent authority takes the final decision about whether the claim is to be compensated and the amount to be paid. The only exception is the compensation of reindeer predation in Sweden, which is paid *a priori* based on the number of reproductions of lynx (*Lynx lynx*) and wolverine (*Gulo gulo*) and on the presence of bear and wolf (*Canis lupus*) (Fourli 1999). Many countries have operated *a posteriori* compensation schemes for more than 30 years and some, for example

France and Slovakia, as long as 50 years. Nevertheless, others such as Estonia have only recently started to compensate damage (Table 1).

PATTERNS OF DAMAGE CLAIMS

In Europe, over 3200 claims for bear damage are compensated per year by the responsible authorities. Overall, we collated records of about 18 300 compensated damage claims from 18 management units across Europe within the period 2005–2012 (Fig. 1, Table S2). The compensated items included, among others, 42 400 sheep, 1500 cattle and almost 11 200 beehives (see Tables S2–S4). Most of the claims corresponded to damage to livestock (59%), followed by claims for beehives and agricultural losses (21% and 17%, respectively). Claims for livestock damage occurred all over Europe, but were less frequent in eastern European countries (e.g. Poland and Estonia). In most of the studied management units, claims for livestock losses primarily involved predation on sheep (Table S5). However, in the Greek Pindos, about 65% were due to cattle losses, representing almost 50% of the total claims for the management unit. The number of sheep per damage claim varied widely across Europe. For instance, in the Polish Western Carpathians, an average of 6·3 sheep were compensated per damage claim (SD ± 3·2), compared to 1·3 (SD ± 1·2) in Estonia. The majority of claims for damaged apiaries occurred in the Mediterranean and eastern European regions. On average, 3·7 beehives (SD ± 1·4) were destroyed per claim (Table S5). Damage to agriculture was mostly claimed in management units in southern Europe and was of considerable importance in the Dinaric–Pindos population (Fig. 1).

We found that the typology of damage claims differed among management units; while in eastern Cantabria almost all claims were for damage to apiaries, in France most were due to livestock depredation and in Slovenia the claims were evenly distributed among damage types (Fig. 1, Table S2).

BEAR DAMAGE RATIO

For calculations of the bear damage ratio, we considered 17 919 claims from 16 management units with similar compensations schemes (Table 2). Croatia and Sweden were excluded due to the incomplete data. In Croatia, a significant portion of the claims were not available since not all hunting associations provided data on compensated claims (Huber *et al.* 2008a). In Sweden, no data were available on the total number of claims for livestock damage because damage to reindeer was under the *a priori* compensation scheme.

The damage ratio varied greatly among management units. The French Pyrenees and the Scandinavian population in Norway showed the highest damage ratio in Europe, with more than 7 compensated claims per bear annually. Estonia had the lowest damage ratio, with < 0·1 claims per bear and year, followed by the Western Carpathians of Poland and Slovakia, with < 0·2 claims per bear and year (Table 2).

Values of the damage ratio varied across management units within the same bear population; for example in the Pyrenean population, the damage ratio was nine times higher in France than in Catalonia (Table 2). It also varied among management units occurring in the same country and, therefore, with the same compensation system and management measures; for example in Poland, the damage ratio for the total number of claims was six times higher in the eastern than in the Western subpopulation. The three units with reintroduced populations had damage ratios twice as high as the remaining management units (4·24 ± 3·31 vs. 2·05 ± 2·56 claims per bear and year; mean ± SD).

LARGE-SCALE FACTORS ASSOCIATED WITH CLAIMS

The management hypothesis had the highest 'hypothesis weight' for all the response variables (range of AIC_c weight: 0·692–0·860; Table 4). Within this hypothesis, supplementary feeding was the most significant explanatory variable and showed a negative relationship to livestock and the total number of damage claims per bear and km^2 (see standardized estimates in Table 4). The univariate model including the effect of supplementary feeding on livestock claims had the highest weight among all competing models (AIC_c weight = 0·61). Harvest also showed a negative effect and was important in explaining the variation in the number of claims for apiary damage (univariate model: AIC_c weight = 0·51). Although reintroduced populations generally had a higher number of associated claims for livestock damage, the univariate model received little support (AIC_c weight = 0·011).

The human land-use hypothesis was second in importance (range of AICc weight: 0·130–0·283). The percentage of agricultural cover showed a negative relationship to the number of damage claims per bear and km^2 (total, livestock and apiary claims). Percentage agricultural cover was the strongest predictor for each response variable (mean standardized estimates ± SE from full models: −2·1 ± 0·44, −1·6 ± 0·57 and −1·7 ± 0·41 for livestock, apiary and total claims, respectively; Table 4). Moreover, it was the only variable with a significant and consistent effect across all responses (see Table 4).

We found almost no quantitative evidence to support the forest availability and the economic hypotheses; forest cover and ecotone, as well as economic indicators, did not explain the variation in the number of claims between management units (AIC_c weights < 0·02 for all response variables and both hypotheses; Table 4). We found a weak positive effect of the GDP.PPP per inhabitant on the apiary and the total number of damage claims (Table 4).

Finally, we found no relationship between the number of bears and the number of damage claims for any of the response variables; thus, we found no evidence to support the bear population size hypothesis (see Table 4).

Table 4. Summary of model selection used to explain the variation in the number of compensated claims for brown bear damage in Europe. Generalized linear mixed models were fitted with the management unit as a random factor and using a negative binomial distribution. Model selection was performed separately for the total number of claims (including damage to livestock, apiaries, agriculture and others), the number of claims for livestock damage and the number of claims for damaged apiaries as response variables. The estimated number of bears and the surface of the management units (both transformed to their natural logarithm) were included as an offset in every model, except for the models testing the bear population size hypothesis that included as offset only the surface of the management unit. The 95% confidence intervals are shown in brackets below the estimates. The AIC_c weight (w_i) indicates the likelihood of a given model or hypothesis. The hypothesis weight was calculated as the sum of the w_i of the hypothesis full model and its nested univariate models. The explanatory variables were standardized, and therefore, the estimates are comparable within responses. Note that the models testing the bear population size hypothesis are not comparable with the models testing the other hypotheses because their offset terms differ

			Livestock claims				Apiary claims				Total claims			
Model	Variables	d.f.	Estimate	AIC_c	Delta	Model/ H.weight	Estimate	AIC_c	Delta	Model/ H. weight	Estimate	AIC_c	Delta	Model/ H.weight
Forest availability H.						0·004				0·006				0·003
Full	Forest cover	5	−0·60 (−1·7, 0·55)	753·1	13·7	0·001	−0·48 (−1·4, 0·43)	728·2	11·8	0·001	−0·49 (−1·5, 0·53)	848·1	13·3	0·001
	Ecotone		0·064 (−1·1, 1·2)				−0·71 (−1·6, 0·20)				−0·34 (−1·4, 0·67)			
Univariate	Forest cover	4	−0·58 (−1·7, 0·53)	750·8	11·4	0·002	−0·37 (−1·3, 0·60)	728·1	11·6	0·002	−0·44 (−1·5. 0·58)	846·3	11·5	0·001
Univariate	Ecotone	4	−0·063 (−1·2, 1·1)	751·8	12·4	0·001	−0·64 (−1·6, 0·30)	726·9	10·5	0·003	−0·28 (−1·3, 0·76)	846·7	11·9	0·001
Human land-use H.						0·130				0·283				0·223
Full	Agricultural cover	5	−2·1* (−3·3, −1·0)	743·5	4·1	0·078	−1·6* (−2·4, −0·79)	719·8	3·4	0·091	−1·7* (−2·6, −0·87)	838·1	3·3	0·081
	Human density		1·1 (−0·057, 2·2)				0·38 (−0·44, 1·2)				0·50 (−0·38, 1·4)			
Univariate	Agricultural cover	4	−1·5* (−2·4, −0·53)	744·5	5·0	0·050	−1·4* (−2·1, −0·69)	718·4	2·0	0·191	−1·5* (−2·2, −0·71)	837·0	2·2	0·141
Univariate	Human density	4	−0·30 (−1·5, 0·90)	751·6	12·2	0·001	−0·49 (−1·5, 0·51)	727·7	11·3	0·002	−0·45 (−1·5, 0·62)	846·3	11·5	0·001
Management H.						0·860				0·692				0·756
Full	Reintroduced	6	0·55 (−0·27, 1·38)	741·5	2·0	0·222	0·16 (−0·55, 0·86)	719·8	3·3	0·095	0·26 (−0·46, 0·98)	836·8	2·0	0·156
	Suppl. feeding		−1·4* (−2·3, −0·56)				−0·43 (−1·4, 0·53)				−0·89 (−1·9, 0·079)			
	Harvest		−0·28 (−1·1, 0·57)				−1·1* (−2·0, −0·19)				−0·73 (−1·6, 0·19)			
Univariate	Reintroduced	4	1·2* (0·16, 2·2)	747·4	8·0	0·011	0·73 (−0·22, 1·7)	726·5	10·1	0·003	0·92 (−0·050, 1·9)	843·9	9·1	0·004
Univariate	Suppl. feeding	4	−1·7* (−2·5, −0·96)	739·4	0·0	0·615	−1·3* (−2·0, −0·56)	720·0	3·5	0·087	−1·5* (−2·2, −0·85)	834·8	0·0	0·418
Univariate	Harvest	4	−1·2* (−2·2, −0·14)	747·6	8·1	0·011	−1·5* (−2·1, −0·81)	716·4	0·0	0·506	−1·5* (−2·2, −0·73)	836·5	1·7	0·177
Economic H.						0·003				0·016				0·015
Full	GDP per km^2	5	−0·11 (−0·96, 0·75)	753·6	14·2	0·001	−0·19 (−1·2, 0·79)	726·5	10·0	0·003	0·046 (−0·88, 0·97)	844·8	10·0	0·003
	GDP per inhab		0·25 (−0·49, 0·98)				0·94 (−0·045, 1·9)				0·67 (−0·23, 1·6)			
Univariate	GDP per km^2	4	0·070 (−0·59, 0·73)	751·8	12·4	0·001	0·47 (−0·30, 1·2)	727·2	10·8	0·002	0·54 (−0·11, 1·2)	844·4	9·6	0·003
Univariate	GDP per inhab	4	0·19 (−0·39, 0·77)	751·4	12·0	0·001	0·81* (0·10, 1·5)	724·3	7·9	0·010	0·70* (0·081, 1·3)	842·5	7·7	0·009
Null	Intercept	3	−10·6* (−11·5, −9·2)	749·7	10·2	0·004	−9·9* (−10·9, −9·0)	726·4	10·0	0·003	−9·0* (−10·0, −7·9)	844·7	9·9	0·003
Bear population size H.														
Full	The number of bears	3	0·11 (−0·59, 0·81)	747·0	2·1	0·26	−0·12 (−0·54, 0·33)	715·2	2·0	0·27	−0·035 (−0·56, 0·48)	837·3	0·0	0·249
Null	Intercept	2	−5·8 (−6·7, −4·9)	744·9	0·0	0·74	−5·4 (−5·9, −4·8)	713·2	0·0	0·73	−4·4 (−5·1, −3·7)	835·1	2·2	0·751

d.f., degrees of freedom; AIC_c, Akaike Information Criteria for small sample size; H., hypothesis.
Explanatory variables as in Table 3.
*Significant effects (estimates excluding zero from the 95% confidence interval).

Discussion

Brown bears in Europe raid beehives more often than in any other continent. However, predation on livestock is the most frequent type of brown bear damage in Europe, while damage to crops and orchards and to garbage bins is more frequent in Asia and North America, respectively (Can *et al.* 2014). This is consistent with our findings that more than half of the claims for bear damage were for livestock losses, followed by damage to apiaries. The availability of, and access to, livestock, apiaries and crops greatly influences the typology and the incidence of damage (Ogada *et al.* 2003; Rigg *et al.* 2011). However, there were no available data to test this association on the European scale.

The number of compensated claims varied considerably among management units (Fig. 1). The Scandinavian population, where bears are claimed to cause considerably more damage on the Norwegian side of the border, is very illustrative. Excluding depredation of free-ranging domestic reindeer, which is compensated *a priori* (and therefore not quantified), in Sweden, farmers have to prove the use of preventive measures in order to receive compensation after claiming for damage. However, in Norway, up to 95% of compensation payments are not verified, and livestock (mainly sheep) is generally free-ranging and unprotected (Swenson & Andrén 2005; Mabille *et al.* 2015). Similar to Sweden, in Croatia and Slovakia compensation is conditional on protection of farming assets, with the aim of reducing both the occurrence of damage and nuisance bears (Huber *et al.* 2008a; Rigg *et al.* 2011). However, the low number of claims in Croatia is also partly due to the dissatisfaction of affected people with the compensated amount and an overly bureaucratic compensation procedure. This suggests that differences in the number of claims among management units are influenced to some extent by the characteristics of individual compensation schemes: they can affect the actual extent of damage through, for example, stimulating the use of prevention measures, as well as the amount of verified damage by influencing the willingness of people to claim damage or, indeed, to make false claims.

The damage ratio widely varied among management units, and we found large differences also within transboundary populations. For instance, in the Rhodope Mountains, the number of damages claimed per bear and year was three times higher in Greece than in Bulgaria. We also found the differences among management units within the same country (e.g. Norway, Poland and Greece; see Table 2). This indicates that the observed variation in damage claims is not solely due to the variation in compensation schemes among countries. We found that human land use and management measures had an effect on the number of damages claimed.

The management hypothesis had the highest support. Supplementary feeding showed a variable effect across responses, which was negative and significant for the livestock and the total number of damage claims; claims were less frequent in units with supplementary feeding (Table 4). A plausible explanation for this result could be that the availability of supplementary food, which is predictable and rather stable, may buffer the variations in the availability of natural foods, which may affect damage occurrence (Gunther *et al.* 2004; Garshelis & Noyce 2008). It is also possible that supplementary feeding is masking other factors not considered in our analysis. For example, supplementary feeding is most common in central and eastern European countries, many of which lack a long tradition of compensation systems, but have a history of coexistence with large predators. Therefore, people in these countries may keep using traditional prevention measures to coexist with large carnivores. Some studies show that the presence of attractants may increase the risk of bear damage at regional scales (Wilson *et al.* 2006; Northrup, Stenhouse & Boyce 2012); however, the existing literature provides mixed evidence about the potential effects of supplementary feeding on bear damage (Kavčič *et al.* 2013). Therefore, we advise caution in the interpretation of our results in relation to supplementary feeding and highlight the need for further research on this topic.

The effect size of bear harvesting varied across the response variables and was important and negatively related to the number of claims for apiary damage (Table 4). This is in agreement with the available scientific literature, which reports variable outcomes of bear hunting (see Treves 2009). Nuisance individuals may cause a disproportionate amount of damage irrespective of population size, and they may be more likely removed in areas where bear hunting is allowed. Hunting might select against those bears that have learnt or inherited an attraction to apiaries, often located close to human settlements (Treves 2009). However, there is no conclusive evidence that carnivore harvesting helps to reduce property damage and conflicts: the reduction in predator density does not always result in decreasing livestock losses (e.g. Treves, Kapp & MacFarland 2010; Wielgus & Peebles 2014) and increases in predator's culling quotas do not necessarily improve people's tolerance towards the hunted species (Treves, Naughton-Treves & Shelley 2013).

Human land use was also important to explain the number of damage claims. Clearly, there were fewer claims for damage caused by bears living in areas with high agricultural cover. A straightforward explanation of this result is that areas with high land-use intensity are less frequented by bears (Fernández *et al.* 2012) and, thus, are less susceptible to damage. In human-dominated landscapes, losses due to predation on livestock are more likely in areas with fewer people (Ogada *et al.* 2003).

We found no association between the number of damage claims and the number of bears, supporting previous findings in Europe (Kaczensky 1999). For instance, while the bear population in Poland and the Apennines has remained stable over the last two decades (see Jakubiec 1990 and Gula, Frackowiak & Perzanowski 1995 for Poland, and Chapron *et al.* 2014 and references therein for the Apennines), livestock depredation rates have decreased in Poland (from an average of 87 livestock losses per year in 1987–91 to 8 in 2005–10) and increased in the Apennines (from an average of 71 livestock losses per year in 1980–88 to 147 in 2005–09 in the Apennines; see Kaczensky 1999 and Table S3). Similarly, comparing our results with those of Swenson & Andrén (2005), we see that in Norway both the number of bears and sheep losses compensated have roughly tripled in a 20-year period (sheep losses: 1998 per year in 1993–95 to 5678 per year in 2005–2010), whereas in Sweden the bear population size has also tripled, but the number of sheep losses has slightly declined (from 98 per year in 1993–95 to 72 per year in 2005–2010). Although a better understanding of these situations requires a more in-depth analysis, the above comparisons illustrate that the variation in the number of damage claims in a given region is not necessarily related to the variation in the size of its bear population.

CONCLUSIONS

This is the first study of wildlife damage that integrates an assessment of the incidence of compensated claims with an analysis of ecological and socio-economic correlates at a continental scale. We showed that the number of claims for bear damage is a complex issue determined by multiple factors, including the functioning of damage compensation schemes, human land use and management practices. Policies that ignore this complexity and focus on a single factor, such as bear population size, may not be effective in reducing claims. The effect that ecological variables, such as forest availability, can have on the number of damage claims at a regional or landscape scale (e.g. Treves *et al.* 2011) seems to be diluted by the stronger effect of human-related factors at the continental scale. We suggest that the reduction in damage claims requires schemes that implement prevention and compensation of damage in parallel, and condition compensation on the application of preventive measures. Effective policies should be based on integrative approaches that prioritize prevention efforts in areas where damage claims are more likely to occur, for example in the case of reintroduced or expanding populations.

This study presents a large amount of information on the compensation systems and bear management from 26 countries in Europe, including a total of 18 300 damage claims. Although some management units were excluded from the statistical analysis due to the incomplete data, all the bear populations in Europe were represented and a variety of environmental and socio-economic conditions covered. Therefore, we stress the applicability of our findings to the whole of Europe. The application of similar approaches in future studies of other wildlife species and in other continents could significantly improve our understanding of conflicts arising from wildlife damage.

Acknowledgements

This study was funded by the National Science Center in Poland under agreement DEC-2013/08/M/NZ9/00469. J.A. was funded by the project GLOBE POL-NOR/198352/85/2013 under the Polish-Norwegian Research Programme operated by the National Centre for Research and Development. N.S. and J.N. designed the study; C.B. compiled and analysed the data collected by the co-authors with inputs from J.A., A.K.S., N.F., N.S., J.N. and E.R.; C.B., N.S. and E.R. wrote a first draft that was improved by J.N., J.A. and N.F. All the authors declare no conflict of interest and critically revised the manuscript. We thank two anonymous reviewers for their helpful comments and Miguel Delibes, Andrés Ordiz, Jon Swenson, Petra Kaczensky, Ernesto Díaz, Leonardo Gentile, Vadim Sidorovich, Ihor Dykyy, Pavlo Hoetsky, George Predoiu, Ilpo Kojola, Marcela Kocianova-Adamcova and Tõnu Talvi for helping in networking and data collection.

Data accessibility

Data from this study are available from Dryad Digital Repository doi:10.5061/dryad.7v11h (Bautista *et al.* 2016).

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Received 31 August 2015; accepted 23 May 2016
Handling Editor: Jacqueline Frair

Supporting Information

Additional Supporting Information may be found in the online version of this article.

Table S1. List of references and data sources.

Table S2. Number of compensated claims for brown bear damage per year in Europe.

Table S3. Number of livestock losses compensated as brown bear damage in Europe.

Table S4. Number of beehives and agriculture losses compensated as brown bear damage in Europe.

Table S5. Number of destroyed beehives and sheep lost per compensated claim for brown bear damage in Europe.

Table S6. Characterization of the management units included in the analysis.