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Human–wildlife conflict: socio–ecological factors influencing livestock predations and agricultural damages

Dottorando

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“If people lose knowledge, sympathy and understanding of the natural world, they’re going to mistreat it and will not ask their politicians to care for it”.

Sir. David Attenborough

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Summary

Wildlife conservation is one of the key points that need to be considered to maintain ecosystem stability. In the same way, extensive livestock practices represent a fundamental traditional and cultural heritage for local people, beyond offering several ecosystem services in the form of food, habitat and biodiversity preservation, and landscape maintenance. This research was conducted with the main purpose to evaluate the magnitude of conflict between wild species (especially large carnivores) and livestock activities mainly in the mountainous areas of the north–east of Italy.

In the first step, we assessed the degree of perception of different stakeholders categories about carnivores' presence and conservation. The first manuscript was then finalized to realize a meta–analysis starting from the scientific manuscripts dealing with human–carnivore (i.e., brown bear *Ursus arctos*, grey wolf *Canis lupus*) conflicts at European Union (EU) level. The findings presented revealed that rural inhabitants and hunters exhibited the most negative attitude towards brown bears and grey wolves compared to urban inhabitants and conservationists, whose attitude was more positive. Furthermore, the direct experience with predators as a consequence of the ongoing re–colonization may have affected the degree of acceptance of certain categories, and the long–term coexistence between humans and carnivores does not necessarily imply increased tolerance. To encourage coexistence, we recommend monitoring changes in attitudes over time relative to carnivore population dynamics.

In the second step, we estimated the abundance of a wild species considered as a pest because of damages to croplands. The second manuscript was then finalized to realize the first estimation of the crested porcupine *Hystrix cristata* abundance in the Tuscany region (central Italy). The choice to concentrate the research activities in central Italy was taken because the crested porcupine is still present at very low densities in the north–east of Italy. The findings presented showed that an average minimum number of 1803 crested porcupines were estimated within an area covering about 17,111 km^2. Capture and removals represent only a temporary solution to reduce crop damages since the area is rapidly re–colonized by other individuals. Moreover, the use of visual deterrents and/or electric fences seems to be poorly effective since crested porcupines get rapidly used to these devices and present hollow quills, which make them immune

to electrical shocks. Tin fences (partly buried) at about 50 cm heigh from the ground seem to be the most effective method to prevent crop damages.

In the third step, we explored the intensity of human–carnivore conflicts in north–eastern Italy to find those areas in which mitigation interventions need to be prioritized. Using data on official claims referring to both brown bear and grey wolf attacks towards livestock, the third manuscript was then finalized to identify 'hotspot' conflictive areas within the Autonomous province of Trento and both the Veneto and Friuli Venezia Giulia regions. The findings obtained revealed that the different feeding behaviour of brown bears and grey wolves, could lead the latter to be more problematic than the firsts in terms of livestock attacks. No 'hotspot' conflictive areas were observed as far as bear predations are concerned. Conversely, for what concerns wolf predations, a 'hotspot' conflictive area was observed in the 'Lessinia' highland where, to date, at least four packs are present and livestock (especially during the last decade) was frequently left free to graze in open pastures, unattended and unprotected during both day and night. Conservationists and carnivore–policy makers should therefore give appropriate support to livestock owners to encourage the implementation of proper prevention measures aimed at fostering the coexistence between large carnivores and humans in the Italian Alps.

The results obtained from this research should be used to delineate efficient education strategies to sensibilize certain stakeholder categories regarding the importance of preserving biodiversity, and to implement effective conservation and management interventions to promote the long–lasting coexistence between wildlife and human activities.

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1. Introduction

The term human–wildlife conflict is generally used to describe the negative interactions occurring between people and wild species (Woodroffe *et al*. 2005) and includes: (*i*) actions pursued by humans or wildlife that produce an adverse effect on the other (Conover 2002), (*ii*) real threats posed by wildlife to human lives, recreation and/or economic security (Treves & Karanth 2003), or (*iii*) the perception that wildlife may constitute a threats for human safety, property, and food (Peterson *et al*. 2011).

The impacts of wildlife on humans can be direct or indirect. Among the direct impacts we have material and economic damages to livestock, crops, game species, and properties (Woodroffe *et al*. 2005; Linnell *et al*. 2010; Guerisoli *et al*. 2017). Indirect impacts are instead more difficult to measure, and include diminished physiological well–being, disruption of livelihoods, and food insecurity (Woodroffe *et al*. 2005; Linnell *et al*. 2010; Dickman *et al*. 2011). The frequency of conflicts can be also highly variable within and among geographic regions, with some properties better protected than others, which in turn lead some households to experience little damages whereas others may experience a surplus of damages (Dickman *et al*. 2011). The most extreme ecological effect derived from conflict with humans is the extinction of wild species. In fact, the absence of appropriate support by conservationists and wildlife–policy makers and/or the scarce acceptance by local stakeholders may lead to wildlife persecution in the form of poaching and/or retaliatory killings (Graham *et al*. 2005; Quiroga *et al*. 2016). Particularly, the persecution and decline of species like large predators have led to detrimental cascading ecological effects on ecosystems (Ripple *et al*. 2014).

Human–wildlife conflicts occur in different ecosystems and involves a large number of rare and protected species (like carnivores), and/or abundant species considered as pests (Nyhus *et al*. 2016). As for carnivores, certain stakeholder categories (e.g., farmers, livestock owners, hunters) generally express deep hostility towards large predators because of the real and/or perceived threats for human safety and livelihoods (Franchini *et al*. 2021). Felids and canids are frequently involved into conflicts with humans because of their large home–ranges, great body mass, and dietary requirements (Macdonald & Sillero-Zubiri 2004; Macdonald & Loveridge 2010). The abundance of predators depends on wild prey availability, and livestock predations may increase if the abundance of the latter is reduced (Khorozyan *et al*. 2015). However, an ecological concept known as 'optimal foraging theory' (Werner & Hall 1974) dictates that, to maximize fitness, an animal adopts a foraging strategy that provides the highest benefits (energy) at the lowest costs. In this sense, even in the presence of high abundance of wild prey, livestock predations could occur especially if livestock are left unprotected or unattended. Despite when talking about human–wildlife conflicts charismatic species like tigers *Panthera tigris* and wolves *Canis lupus* receive notable attention, also other species are amongst the most conflictual ones. Specifically, agricultural pests (i.e., animals that are considered as dangerous to livestock and/or crops) (Waterfield & Zilberman 2012) can cause substantial agricultural damages. A common pest management approach is to disperse and/or eradicate as many individuals of a species as possible. For instance, in 2014 the US Department of Agriculture's Wildlife Service dispersed and killed about 28 million and 2.7 million animals, respectively (Nyhus *et al*. 2016). However, its implementation may vary depending on local legislations.

Despite human interactions with wildlife (and vice–versa) are often framed negatively, several benefits also exist in the form of educational and/or recreational activities, as well as ecosystem services (ESs) provided to the community (Soulsbury & White 2015). For instance, a study conducted in Mumbai, India (Braczkowski *et al*. 2018) showed that common leopards *Panthera pardus*, through direct predation on stray dogs *C. l. familiaris*, reduced considerably the spread of diseases (e.g., rabies) among humans, thus providing important services in terms of public health benefits.

Some of the most important drivers of conflicts between humans and wildlife include the human population grow and associated increase of technology, land use and livestock practices (Nyhus *et al*. 2016). Extensive livestock practices are multifunctional and provide important ESs to the society (e.g., Bernués *et al*. 2011; Battaglini *et al*. 2014). ESs are defined as the contributions (direct or indirect) of ecosystems to humans and are classified into four main groups: (*i*) supporting ESs (e.g., photosynthesis, nutrient cycling), (*ii*) provisioning ESs (e.g., food, water, timber), (*iii*) regulating ESs (e.g., climate regulation, water purification), and (*iv*) cultural ESs (e.g., recreational, cultural and spiritual benefits) (Pachoud *et al*. 2020). Extensive grazing practices carried out in mountainous environments contribute to the maintenance of semi–natural grasslands and, at the same time, provide multiple ESs like supporting ESs (e.g., soil carbon sequestration, protection from landslides and fires, conservation of natural habitats and biodiversity) (e.g., Laiolo *et al*. 2004; Burrascano *et al*. 2016; Silva *et al*. 2019), provisioning ESs (e.g., production of milk and traditional cheese) (Bernués *et al*. 2011; Battaglini *et al*. 2014), and also cultural ESs (e.g., touristic attractiveness of the landscape, conservation of the cultural heritage) (e.g., Scolozzi *et al*. 2015; Schirpke *et al*. 2019). Therefore, the amelioration and mitigation of human–wildlife conflicts are central to enhance biodiversity preservation and restoration and, at the same time, the maintenance of traditional grazing practices, especially in mountainous areas.

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2. Purposes of the research

Given the need to increase the awareness of people about the importance to conserve wild species to prevent the disruption of the ecosystems and the role that extensive grazing practices have in terms of ecosystem services provided, biodiversity preservation and landscape maintenance, the main purposes of this research were:

1. To assess the degree of perception of different stakeholders categories about carnivores' presence and conservation. More precisely, the first manuscript aimed to realize a meta–analysis starting from the scientific researches dealing with human–carnivore (i.e., brown bear *Ursus arctos*, grey wolf *Canis lupus*) conflicts realized within the European Union (EU) community.
2. To estimate the abundance of a wild species considered as a pest because of damages to croplands to delineate adequate management and conservation strategies. More precisely, the second manuscript aimed to realize the first estimation of the crested porcupine *Hystrix cristata* abundance in the Tuscany region (central Italy). This species, despite the low damages caused to agriculture, is widely considered as one of the main agricultural pests by farmers. The choice to concentrate the research activities in central Italy was taken because the crested porcupine is still present at very low densities in the north–east of Italy.
3. To assess the intensity of human–carnivore conflicts in north–eastern Italy to find those areas in which mitigation interventions should be prioritized. More precisely, using data on official claims referring to both brown bear and grey wolf attacks towards livestock, the third manuscript aimed to identify 'hotspot' conflictive areas within the Autonomous province of Trento and both the Veneto and Friuli Venezia Giulia regions, areas considered as conflictive because of the presence of large carnivores and extensive grazing activities realized in mountainous areas.

3. The Return of Large Carnivores and Extensive Farming Systems: A Review of Stakeholders' Perception at an EU Level

Original manuscript: Franchini, M.; Corazzin, M.; Bovolenta, S.; Filacorda, S. The Return of Large Carnivores and Extensive Farming Systems: A Review of Stakeholders' Perception at an EU Level. Animals 2021, 11, 1735. https://doi.org/doi.org/10.3390/ani11061735

3.1 Introduction

Extensive grazing practices are less modified and more biodiverse than intensive livestock systems [1] and play a fundamental role in both the management and the conservation of areas of high natural value since these are important providers of ecosystem services (e.g., food, climate regulation, habitat and biodiversity maintenance, etc.) that contribute to human well-being [2,3]. Their importance in production, environmental, and social terms is recognized by European agricultural policies, which also provide direct payments for the public goods offered to society [4]. In the second half of the 20th century, drivers such as urban expansion [5], the shift towards tourism [6], agricultural intensification in the lowlands supported by institutional reforms (e.g., Common Agricultural Policy) [7], and land abandonment in less-favoured areas [8] markedly reshaped the traditional agricultural landscapes [7]. Traditional extensive livestock systems of Europe's mountainous areas have been particularly affected [9] with subsequent abandonment, mainly of upland and less productive areas [10]. These re-naturalization phenomena have favoured the return of shrub and arboreal vegetation [11] and, consequently, of wildlife species including large carnivores [12]. In areas where human activities continue, wildlife and livestock activities overlap geographically, and such co-occurrence may lead either to positive or negative interactions [13,14]. At a global level, human–wildlife interactions arise for several reasons including progressive human advancement into wilderness areas [15], wildlife population range expansion [16], and wildlife recovery because of successful conservation plans [17]. Large carnivores are the most conflictual species that might exert a negative impact on human activities [18,19]. The existing conflict of interest between carnivore conservation and extensive grazing practices elicits strong emotional responses that may undermine both carnivore survival and the long-term maintenance of traditional husbandry practices [19–24]. Apex predators exert a key role in the maintenance of ecological balance as a consequence of trophic cascade effects [25,26]. Such top-down effects in fact play a major role in regulating ecosystem structure because of both direct (density-mediated) and indirect (behaviourally mediated) impacts on herbivores and other medium-sized carnivore species [25,26]. On the other hand, extensive livestock systems are multifunctional as they provide food and raw materials (e.g., water, fodder, wood, etc.) [27], support for human health through climate regulation, medical plants, and the prevention of soil erosion [28], as well as recreational or cultural activities [29]. Therefore, implementing effective mitigation measures in conflict hot-spots assumes remarkable importance in the preservation of carnivore populations while, on the other hand, fostering the maintenance of traditional husbandry practices.

3.1.1 Theoretical Framework: The Perceptions towards Bears and Wolves

Brown bear (*Ursus arctos*) and grey wolf (*Canis lupus*) (hereafter carnivores) are controversial species that have returned to occupy part of their historical distribution range and are now legally protected in most European countries [30]. The recent return of such predators has evoked several emotions varying from admiration to a desire for their extirpation [31–33]. Several factors may be involved in the perception of carnivores. Firstly, sex, age, and education may play a key role in fostering positive attitudes. In general women, elderly people, people with a lower level of education and less knowledge of the target species show less tolerance [34–36]. Secondly, folklore referring to oral traditions, folk tales, culturally transmitted fear, distaste or love towards certain groups of animals may lead to important conservation issues as some species may survive to the detriment of others [37–39]. Thirdly, people living in rural areas are generally less tolerant than urban inhabitants [35,36,40–42], which in turn is linked to another factor that may drive people's attitudes, i.e., direct experience with carnivores [34,36]. Urban interests are perceived as the dominating norm in society driving political processes and controlling policymaking processes [43]. Contrariwise, as far as political power is concerned, rural inhabitants are perceived to be at a lower level than urban ones [44]. This perception of political subordination is even more clear in relation to carnivore management and creates a situation in which rural people perceive that they are not considered, taken seriously, or given enough participation in the carnivore policy processes [45]. All these situations contribute to generating political alienation in terms of a general mistrust that rural inhabitants have towards actors and institutions of the political system [39,40]. All these situations may promote illegal killing of carnivores with common silence and appraisal by local stakeholders who see hunting violators as defenders of human safety [46,47]. On the basis of the information collected, the purpose of this study was thus to provide an initial comprehensive assessment of the level of acceptance of different stakeholder categories (with special focus dedicated to farmers and livestock owners) in relation to bear and wolf presence within the European Union (EU).

3.2 Materials and Methods

3.2.1 Literature Search

To answer these questions, from September to December 2020, we retrieved peer-reviewed English language scientific material published between 1990 and September 2020 addressing human–wolf/bear conflicts in EU countries. To do so, we used three comprehensive databases (Scopus, Web of Science, Pubmed). Only those countries that have a permanent and reproductive presence of at least one species or are affected by the occasional presence of a given species (e.g., animals in dispersal) without reproduction [12,48–50] (i.e., Austria, Belgium, Bulgaria, Croatia, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Italy, Latvia, Lithuania, Netherlands, Poland, Portugal, Romania, Slovakia, Slovenia, Spain and Sweden) were included in the analysis. Likewise, those countries in which the presence of such predators (neither stable nor sporadic) has not been reported in recent years (i.e., Cyprus, Ireland, Luxemburg, Malta) [12,48–50] were excluded. The United Kingdom, despite being part of the EU up to 2019, was excluded from the analysis as these carnivores were eradicated there well before 1990 [12]. The literature search was carried out using the following research string: *conflict* OR attack* OR predation OR* *damage* AND management OR retaliation OR kill* OR poach* OR mortality OR cull* OR control OR mitigation OR prevention OR attitude OR perception OR compensation AND wild* OR predator OR carnivore* OR bear* OR wolf OR wolves AND zootechny* OR husbandry OR transhumance OR extensive OR graze* OR rural OR rangeland OR farm* OR pasture* OR livestock OR cattle OR sheep OR goat**. The inclusion of "*" was made to include all the possible variations of the word considered (e.g., *cull* cull, culling, culled*). With this screening, we were left with 5040 publications (1217 from Scopus; 2959 from Web of Science; 864 from Pubmed). We then manually screened the remaining publications to identify studies that dealt with depredation on livestock species (cattle, sheep, goats) to evaluate stakeholders' attitude towards both wolves and bears. In those studies (e.g., Swedish ones) in which the attitude towards a larger range of predators (e.g., bear, wolf, lynx, wolverine) was evaluated, we considered only data referring to bears and wolves. The same criteria were used for the different husbandry species (e.g., sheep, reindeer) mentioned in the research. In such cases, only information related to the target livestock species was included in the research (i.e., sheep). In those studies in which stakeholders' attitude was evaluated in more countries all belonging to the EU (e.g., Italy and Greece), we discerned the information reported in both areas (i.e., attitudes for Italy and attitudes for Greece). On the contrary, in those studies in which the attitude of the different stakeholder categories was evaluated in more countries not all belonging to the EU (e.g., Norway and Sweden), we considered only the information reported in EU countries (i.e., Sweden). After removal of duplicates, articles that included either livestock or carnivore species not pertinent to the present research and articles whose topic was out of our scope (e.g., those dealing with human–carnivore conflicts but not referring to stakeholder perceptions), the potential sample was reduced to 40 pieces of scientific research (Table 1; Figure 1). Countries included in the review are shown in Figure 2.

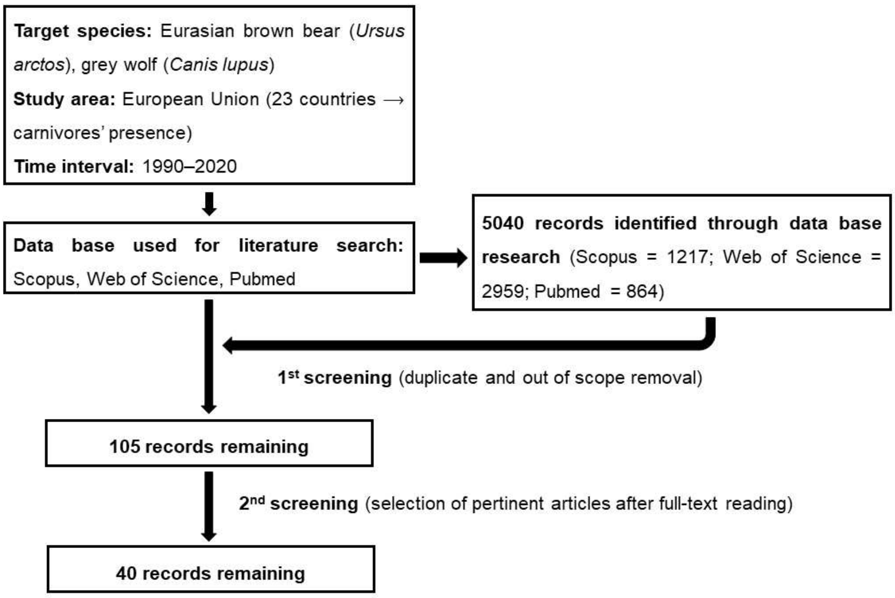


Figure 1. Overview of the criteria used for manuscript selection and dataset creation.

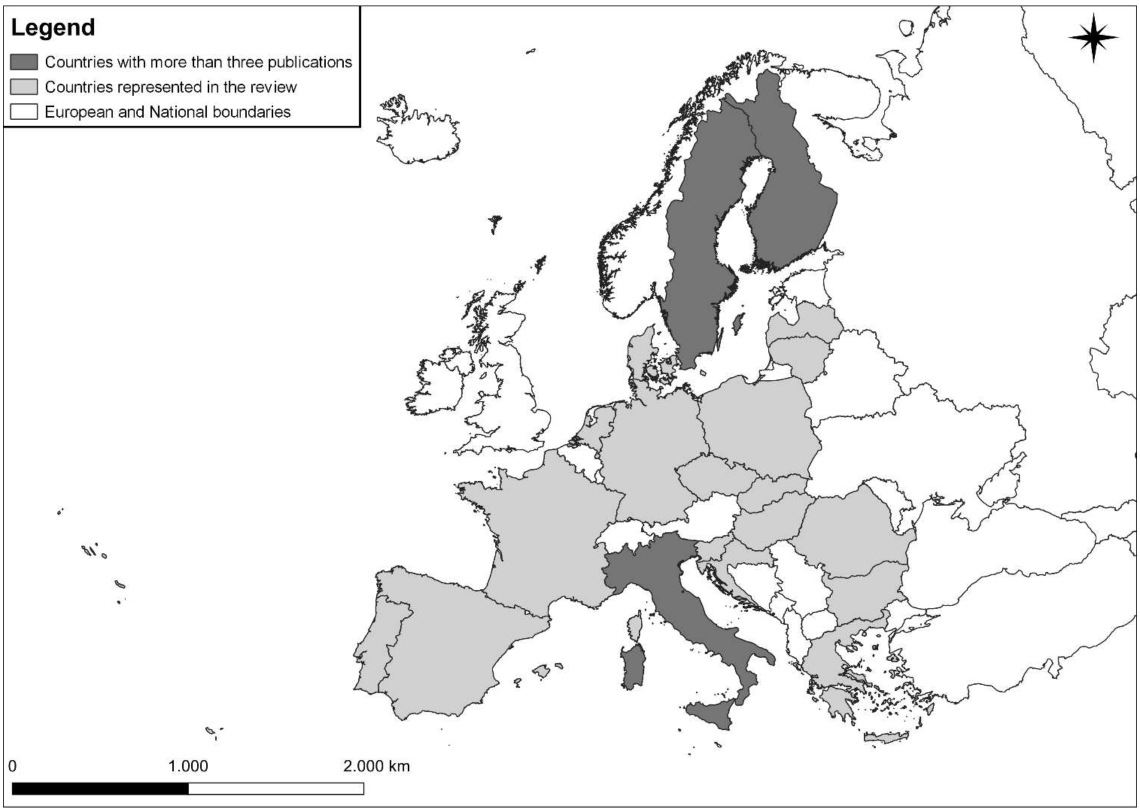


Figure 2. Countries represented in the review referring to the local perception of the involved categories (urban inhabitants, rural inhabitants, hunters, general public, conservationists). In Italy, we found five papers focused on the evaluation of stakeholders' perception towards carnivores. In Sweden, ten scientific articles focused on stakeholders' perception. As far as Finland is concerned, only four articles focusing on the attitude of stakeholders were found.

Table 1. List of publications used in the meta-analysis divided by period and country/ies.

Country/ies	Period	
	2003–2012	2013–2020
Czech Republic	-	[51,52]
Denmark	-	[53]
Finland	-	[46,47,54,55]
France	[56]	[57]
Germany	-	[58]
Hungary	-	[59]
Italy	-	[60–63]
Italy, Greece	[64]	-
Latvia, Lithuania, Bulgaria, Turkey	-	[35]
Lithuania	[40]	-
Netherlands	-	[34]
Norway, France	[65]	-
Norway, Sweden	-	[39,66]
Poland	-	[22,67]
Portugal	[68]	[36]
Romania	-	[69,70]
Slovakia	[71]	-
Slovakia, Romania, Croatia	-	[38]
Slovenia	[72]	-
Spain, Portugal	[37]	-
Sweden	[73–77]	[23,41,42]

3.2.2 Literature Content Analysis

To assess the attitude towards bears and wolves, we performed a meta-analysis using information obtained by standardized questionnaires, a 3- or 5-point Likert-scale, or by using the information reported in each scientific paper (e.g., livestock owners stated that they have low tolerance towards large carnivores or are in favour of the local eradication of the species). Attitudes were standardized as negative, neutral or positive based on the respective Likert-scaled points used in each questionnaire. When considering the 5-point

Likert-scale, attitudes were considered negative if the answer included points 1 and 2, neutral for point 3, and positive for points 4 and 5. The same criteria were used for the 3-point Likert-scale (i.e., 1 = negative; 2 = neutral; 3 = positive). When the attitude was reported as "nuanced" or "variable", depending on specific situations (e.g., direct experiences with the target species) we considered it as a neutral in the analysis. When the attitude varied between neutral and lower, a negative attitude was considered and vice-versa for attitudes varying from neutral to positive. The sampled groups were classified as follows: urban inhabitants, rural inhabitants (mainly farmers and livestock owners), hunters, general public (pet dog owners, guesthouse owners, local educators, berry and mushroom pickers, hikers, fishers, hotel employers, teachers, housewives, pensioners, employees, students), and conservationists (scientists, environmentalists, nature conservationists, non-governmental organizations, park members, foresters). As far as the general public and/or rural inhabitants were concerned, in some cases, some of them were also hunters. Therefore, the attitude reported was divided between the two categories (e.g., if rural people who were also hunters expressed a negative attitude, in the analysis a negative attitude was considered for both rural people and hunters).

To calculate the attitude of each category, the following index was used:

$$A_i = ((x1 * k1) + (x2 * k2) + (x3 * k3))/n$$

where:

A_i = attitude of the i-th category

$x1$ = number of cases in which a negative attitude was mentioned

$k1$ = 1 (value arbitrarily defined for a negative attitude)

$x2$ = number of cases in which a neutral attitude was mentioned

$k2$ = 2 (value arbitrarily defined for a neutral attitude)

$x3$ = number of cases in which a positive attitude was mentioned

$k3 = 3$ (value arbitrarily defined for a positive attitude)

$n = x1 + x2 + x3$

The choice to use such an index was taken in order to obtain a comprehensive measure of the attitude of each stakeholder category, taking into consideration the number of times in which a specific attitude was reported by each category and the overall number of attitudes (positive, neutral or negative) reported.

To compare the attitude between areas in which coexistence had ever persisted with those in which carnivores had been eradicated, because of the limited information available for the other stakeholders, we referred only to those categories that showed the most negative attitude, i.e., rural inhabitants and hunters (see *Results*), and included only studies in which information regarding the status and distribution of both bears and wolves across the study area was provided. To answer this question, the analysis was conducted on a small scale (i.e., considering only the small area/s in which the study was carried out) and considering each case as independent. For instance, for Italy, we included two comparative studies: one carried out in the central Alps (where bears were almost totally eradicated) [60] and another in the central Apennines (where humans and bears have coexisted for centuries) [61] and both were considered as independent cases. This choice was made because in some countries (such as Italy, for example), carnivore eradication took place at a local scale and not throughout the country. Therefore, if we had considered the whole country as an area in which carnivores had persisted and/or were totally eradicated, we would have made a conceptual mistake.

3.2.3 Statistical Analysis

Statistical analyses were performed using R Software (version 4.0), and the alpha value was set at 0.05. To test the difference in terms of attitude between categories, Fisher's test [78] was applied, using the number of times (reported as n in the *Results* Section) in which a certain attitude was reported by the i-*th* category as an observed frequency. To best cope with the reduced sample size, comparison between categories (e.g., positive vs. neutral, positive vs. negative, neutral vs. negative) was done using the pairwise nominal independence function through the R package *rcompanion* [79]. Because of the diversity in terms of publications among years and considering the time interval in which such publications were produced, i.e., 14 years (2003, 2004, 2006, 2008, 2009, 2011, 2012, 2013, 2014, 2015, 2016, 2017, 2019, 2020), we performed a subdivision between publications published between 2003 and 2012 (hereafter, the first period, n = 13) and those from 2013 to 2020 (hereafter, the second period, n = 27). This was done to realize a proper comparison between two-time periods of seven years each. Fisher's test was further used to compare attitude variations between periods and between areas in which carnivores were eradicated (before then starting to recolonize their former range) and those in which rural inhabitants/hunters and carnivores had ever coexisted. Comparisons in terms of the attitude index between categories were realized using the non-parametric Kruskal–Wallis H test [80]. The same test was used to compare attitudes between periods and both the coexistence and non-coexistence areas.

3.3 Results

3.3.1 General Attitude towards Carnivores

From the comparison in terms of stakeholders' attitude towards carnivores, we found that urban inhabitants showed no differences between negative ($n = 2$, 22%) and neutral attitudes ($n = 1$, 11%), while a significant difference was obtained between negative and positive attitudes ($n = 9$, 67%) (F-test, $p = 0.02$), as well as between a neutral and positive attitude. Therefore, as reported by the attitude index ($A_{urban} = 2.44$), urban inhabitants showed, in general, a more positive attitude. As far as rural inhabitants were concerned, a significant difference was found between a negative ($n = 27$, 73%) and neutral attitude ($n = 6$, 16%) (F-test, $p < 0.001$) as well as between a negative and a positive attitude ($n = 4$, 11%) (F-test, $p < 0.001$) and between a neutral and a positive attitude (F-test, $p < 0.001$). Hence, as confirmed by the attitude index ($A_{rural} = 1.38$), rural inhabitants revealed a more negative attitude. A similar trend was observed for hunters. A significant difference was noted between a negative ($n = 14$, 78%) and a neutral attitude ($n = 1$, 6%), between negative and positive attitudes ($n = 3$, 17%) (F-test, $p < 0.001$), and between a neutral and a positive attitude (F-test, $p < 0.001$). Nevertheless, hunters ($A_{hunters} = 1.39$) exhibited a slightly less negative attitude than rural inhabitants. As far as the general public was concerned, Fisher's test reported no significant difference among attitudes ($n_{negative}$ = 8, 33%; $n_{neutral}$ = 5, 21%; $n_{positive}$ = 11, 47%) (F-test, $p = 0.11$). This means that, as confirmed by the attitude index (A_{public} = 2.13), the general public generally exhibited a neutral attitude. Conservationists was the category that revealed the strongest positive attitude ($A_{conservationists}$ = 2.64). A significant difference was in fact obtained between a negative (n = 1, 7%) and a neutral attitude (n = 3, 21%) (F-test, $p < 0.001$), between a negative and positive attitude (n = 10, 71%), and between a neutral and a positive one (F-test, $p < 0.001$). No significant difference was found comparing the attitude indexes between categories (KW-test, $\chi 2 = 4$, $p = 0.41$). The frequency distribution of the answers reported is shown in Figure 3.

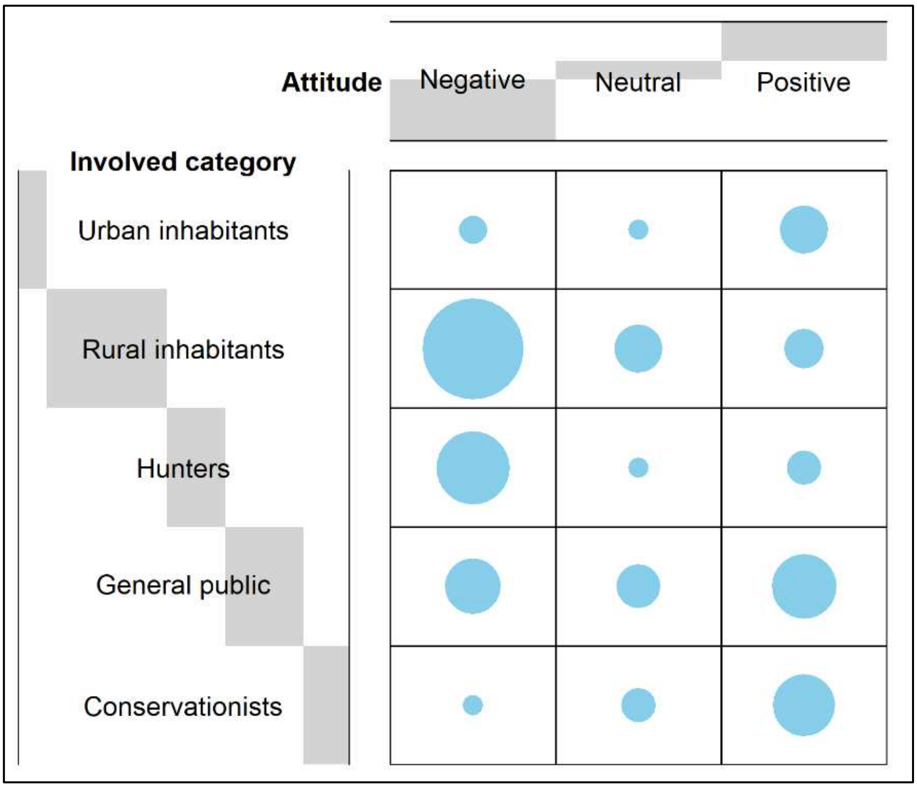


Figure 3. Contingency table showing the distribution frequency as far as the attitude towards carnivores of each category involved is concerned, i.e., urban inhabitants, rural inhabitants (mainly farmers and livestock owners), hunters, general public, conservationists. The size of the grey bars depends on the number of

responses obtained by each stakeholder category for each attitude. For instance, from the above figure we see that the grey bar for rural inhabitants is the largest. This is because the highest number of responses were obtained (negative = 27, neutral = 6, positive = 4). On the contrary, the grey bar for urban inhabitants is the smallest as the lowest number of responses were obtained (negative = 2, neutral = 1, positive = 6). As far as the attitude is concerned (i.e., negative, neutral, positive), the criterion is the same. The negative attitude is that one which showed the larger grey bar as was that one mentioned the most by all stakeholders (urban inhabitants = 2, rural inhabitants = 27, hunters = 14, general public = 8, conservationists = 1). Contrarywise, the neutral attitude showed the smallest grey bar as the lesser mentioned by the stakeholders (urban inhabitants = 1, rural inhabitants = 6, hunters = 1, general public = 5, conservationists = 3). Reference list: [22,23,34–42,46,47,51–77].

3.3.2 Attitude Comparison between Periods

The attitude of urban inhabitants did not change markedly from the first ($n_{negative}$ = 2, 40%; $n_{neutral}$ = 0, 0%; $n_{positive}$ = 3, 60%) to second period ($n_{negative}$ = 1, 25%; $n_{neutral}$ = 1, 25%; $n_{positive}$ = 2, 50%). Despite showing a neutral attitude, the attitude index during the second period (A_{urban} = 2.25) was slightly higher than that shown during the first (A_{urban} = 2.20) firstly because, in the initial period, we did not encounter cases in which a neutral attitude was reported ($n_{neutral}$ = 0) and, secondly, following the formula above, in the first period the number of cases in which attitudes were reported was higher (n = 5) than in the second (n = 4). Furthermore, no significant difference was found in terms of attitude index between periods (KW-test, $\chi 2$ = 1, p = 0.32).

The attitude of rural inhabitants remained negative (F-test, $p < 0.001$) in both the first ($n_{negative}$ = 9, 70%; $n_{neutral}$ = 2, 15%; $n_{positive}$ = 2, 15%) and second periods ($n_{negative}$ = 21, 84%; $n_{neutral}$ = 3, 12%; $n_{positive}$ = 1, 4%), but in the second (A_{rural} = 1.20), this was even more negative than that in the first (A_{rural} = 1.46). Nevertheless, no significant difference was found between periods (KW-test, $\chi 2$ = 1, p = 0.32).

The attitude of hunters changed from neutral (F-test, p = 1.00) in the initial period ($n_{negative}$ = 3, 50%; $n_{neutral}$ = 0, 0%; $n_{positive}$ = 3, 50%) to negative (F-test, $p < 0.001$) in the second, where only negative responses were recorded ($n_{negative}$ = 11, 100%; $n_{neutral}$ = 0, 0%; $n_{positive}$ = 0, 0%). This trend was also confirmed by the attitude index values recorded in both the first ($A_{hunters}$ = 2.00) and second periods ($A_{hunters}$ = 1.00). However, no significant difference was found in terms of the attitude index between periods (KW-test, $\chi 2 = 1$, $p = 0.32$).

The attitude of the general public changed from positive (F-test, $p < 0.01$) in the first period ($n_{negative}$ = 3, 21%; $n_{neutral}$ = 3, 21%; $n_{positive}$ = 8, 57%) to neutral (F-test, $p = 0.98$) in the second ($n_{negative}$ = 4, 29%; $n_{neutral}$ = 5, 36%; $n_{positive}$ = 5, 36%) as confirmed by the respective attitude index values (A_{public} = 2.36 and A_{public} = 2.07 for the first and second periods, respectively). No significant difference was found in terms of the attitude index between periods (KW-test, $\chi 2 = 1$, $p = 0.32$).

The attitude of conservationists remained positive (F-test, $p < 0.001$) in both the first ($n_{negative}$ = 0, 0%; $n_{neutral}$ = 0, 0%; $n_{positive}$ = 4, 100%) and second period ($n_{negative}$ = 1, 11%; $n_{neutral}$ = 2, 22%; $n_{positive}$ = 6, 67%) even though in the first period this was totally positive ($A_{conservationists}$ = 3.00) compared to the second, in which negative ($n = 1$) and neutral ($n = 2$) attitudes were reported ($A_{conservationists}$ = 2.56). No significant difference was found in terms of the attitude index between periods (KW-test, $\chi 2 = 1$, $p = 0.32$).

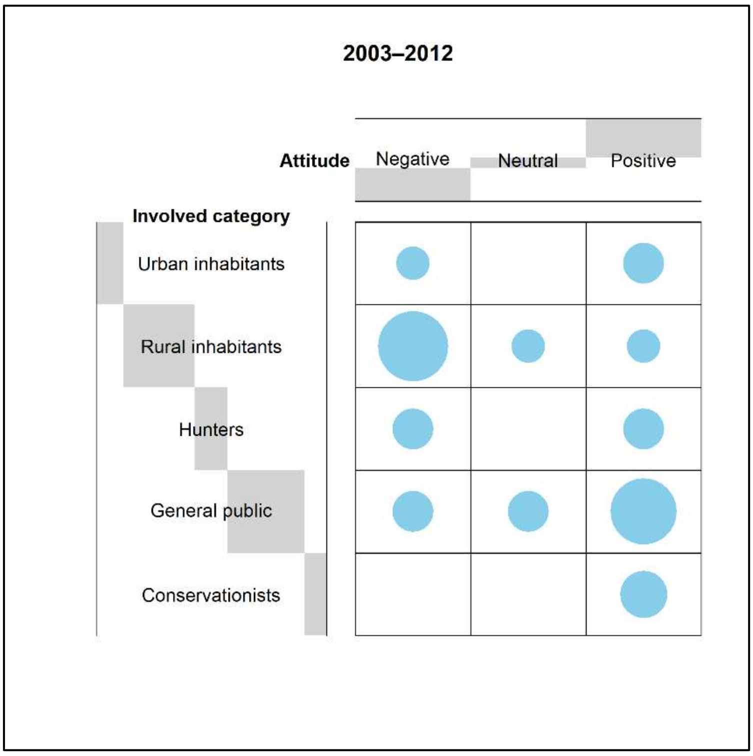
The comparison in terms of frequency distribution of the responses obtained between periods is shown in Figure 4a,b.

3.3.3 A Comparison of Rural Inhabitants' and Hunters' Attitudes between Coexistence and Non-Coexistence Areas

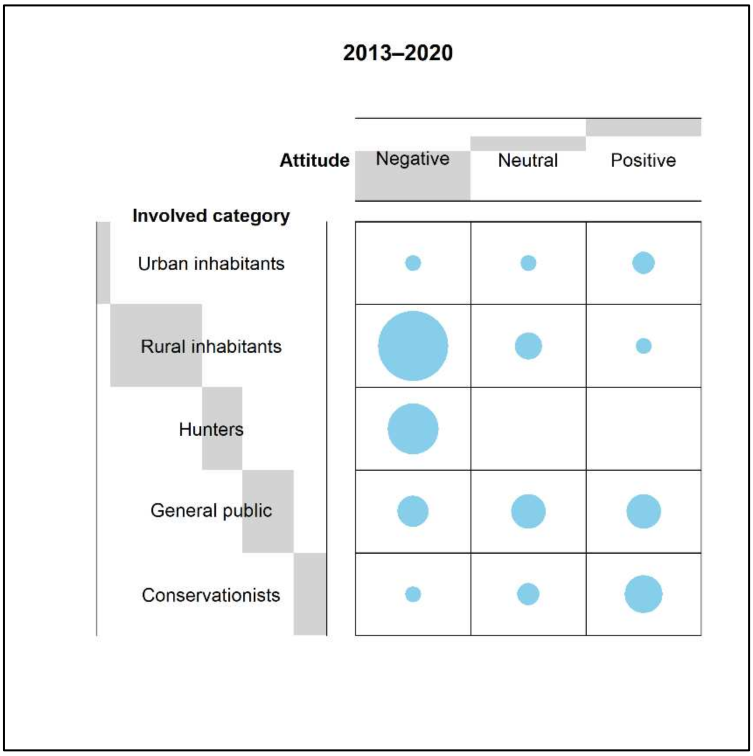
Comparing the attitude of rural inhabitants between areas in which humans and carnivores have always coexisted and the areas in which these carnivores were eradicated, we observed that the attitude was significantly negative in both coexistence ($n_{negative}$ = 2, 67%; $n_{neutral}$ = 1, 33%; $n_{positive}$ = 0, 0%) and non-coexistence areas ($n_{negative}$ = 12, 86%; $n_{neutral}$ = 1, 7%; $n_{positive}$ = 1, 7%). However, in areas where carnivores had been eradicated, this was even more negative (F-test, $p < 0.001$, A_{rural} = 1.21) than in areas where carnivores and humans have coexisted for centuries (F-test, $p < 0.001$, A_{rural} = 1.33). Nevertheless, we did not find a significant difference in terms of attitude indexes between areas (KW-test, $\chi 2 = 1$, $p = 0.32$).

As far as hunters are concerned, the attitude was significantly negative in both coexistence ($n_{negative}$ = 1, 100%; $n_{neutral}$ = 0, 0%; $n_{positive}$ = 0, 0%) and non-coexistence areas ($n_{negative}$ = 7, 70%; $n_{neutral}$ = 0, 0%; $n_{positive}$

= 3, 30%). Nevertheless, even in this case, in areas where carnivores had been extirpated, the attitude was even more negative (F-test, $p < 0.001$, $A_{hunters} = 1.60$) than in areas where carnivores and humans have always coexisted (F-test, $p = 0.02$, $A_{hunters} = 1.00$). However, it is important to specify that the lower attitude index value in the coexistence area is linked to the only negative response obtained. Furthermore, even in this case, no significant difference was found in terms of the attitude index between areas (KW-test, $\chi2 = 1$, $p = 0.32$).



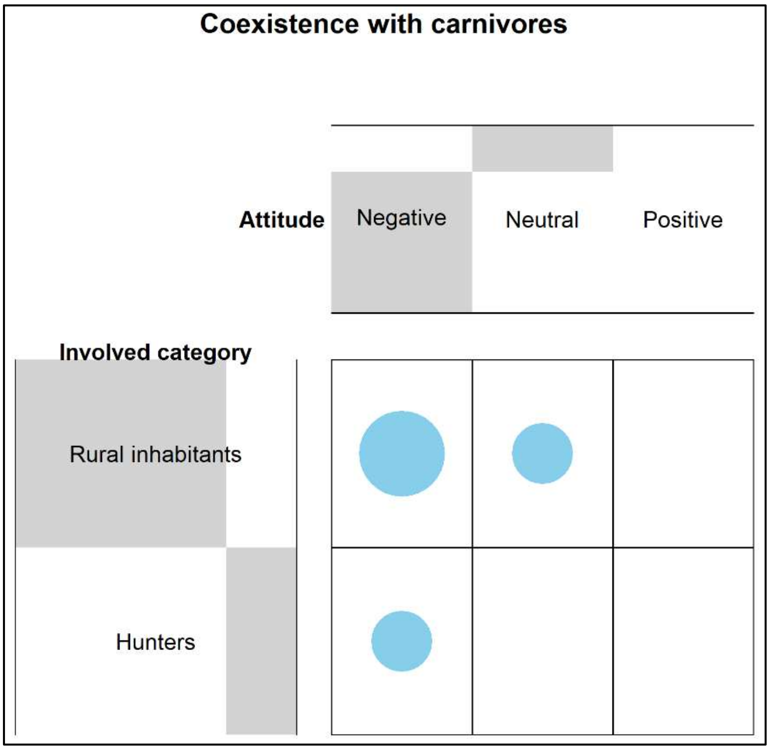
(a)



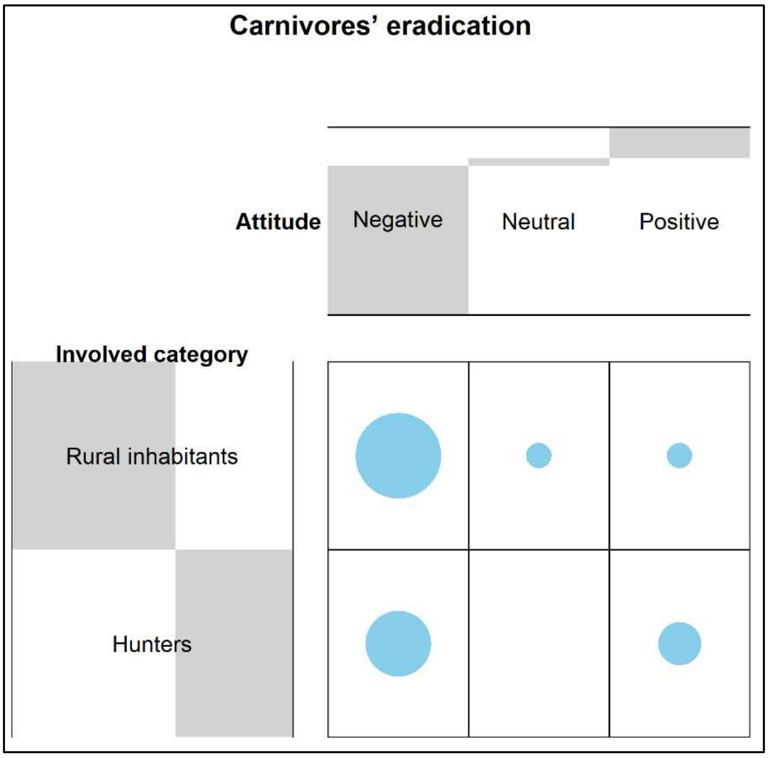
(b)

Figure 4. (**a**). Contingency table showing the distribution frequency regarding the attitude towards carnivores of each category involved in the first period (2003–2012). The size of the grey bars depends on the number of responses obtained by each stakeholder category for each attitude (refer to Figure 3 caption for a more detailed explanation). For the reference list divided by periods, refer to Table 1. (**b**). Contingency table showing the distribution frequency regarding the attitude towards carnivores of each category involved in the second period (2013–2020). The size of the grey bars depends on the number of responses obtained by each stakeholder category for each attitude (refer to Figure 3 caption for a more detailed explanation). For the reference list divided by periods, refer to Table 1.

The comparison in terms of the frequency distribution of the responses obtained between areas is shown in Figure 5a,b.


Coexistence with carnivores
Attitude
Negative
Neutral
Positive
Involved category
Rural inhabitants
Hunters

(a)



(b)

Figure 5. (**a**). Contingency table showing the distribution frequency regarding the attitude towards carnivores of rural inhabitants (mainly farmers and livestock owners) and hunters in areas where humans and carnivores have always coexisted. The size of the grey bars depends on the number of responses obtained by each stakeholder category for each attitude (refer to Figure 3 caption for a more detailed explanation). Reference list: [36,61,69,70]. (**b**) Contingency table showing the distribution frequency regarding the attitude towards carnivores of rural inhabitants (mainly farmers and livestock owners) and hunters in areas where carnivores have been eradicated. The size of the grey bars depends on the number of responses obtained by each stakeholder category for each attitude (refer to Figure 3 caption for a more detailed explanation). Reference list: [34,39–42,52–54,57,59,60,66,68,72–74].

3.4 Discussion

Our study represents one of the first attempts to provide a comprehensive understanding of the attitudes of various stakeholder categories towards bears and wolves at an EU level. From our study, we found that both hunters' and rural inhabitants' attitudes were strongly negative, while urban inhabitants and conservationists were more tolerant. The general public was the only category showing a neutral attitude.

Attitude towards carnivores may vary according to different cultural, socio–economic, and/or political circumstances. This in turn can be linked to socio–economical parameters, history, and wildlife management policies [57,81–84]. Factors such as age [85], sex [86], educational level [87], and direct experiences with target predator species [57,63] seem to exert a key role in driving people's attitudes. For instance, Stauder et al. (2020) [63] showed that urban people living in areas where wolves were absent were more tolerant toward carnivores. By the same token, Piédallu et al. (2016) [57] showed that people living in no-bear areas showed more tolerance than those living in areas with bears. Rural inhabitants, especially farmers and livestock owners, represent the most affected category since the carnivores' return is linked to increased clashes with extensive grazing practices, potentially generating conflicts amongst a range of categories. Indeed, the perceived asymmetry in terms of political power between urban and rural areas may even lead to the development of conflicts between urban and rural inhabitants as the latter feel excluded from the political system [41,42,44]. In the case of policy towards carnivores, livestock owners may view the effective (or perceived) reintroduction of carnivores as a sort of political oppression by urban groups [88]. Moreover, they perceive the presence of predators as a negative factor, potentially limiting their day-to-day activities [89]. Opposing the return of carnivores is then often seen as a necessity, both in the sense of maintaining traditional rural practices and defending their political autonomy in the face of urban interests [88].

We found that the attitude of the categories involved changed between periods (2003–2012 and 2013–2020), except for the general public whose attitudes remained neutral and conservationists whose attitudes remained positive. Indeed, the latter perceived the return of such predators as a positive point to re-establish and maintain ecological balance and to prevent the disruption of ecological systems [25,26]. The attitude of rural inhabitants remained negative in both periods, becoming even more negative during the second, while hunters' attitude changed from neutral in the first period to negative in the second one. Urban people showed a positive attitude during the first period changing to neutral during the second. We interpreted such changes as being that urban residents find it easier to support the large carnivore's return and conservation in areas where they had been eradicated because they did not experience interactions with them [57,63]. Indeed, these results are in line with findings presented by Dressel et al. (2015) [33], who compared the attitude of peoples towards both bears and wolves in Europe from 1976 to 2012.

As expected, the attitudes of rural inhabitants remained negative in both periods as they were more involved in the conflicts. Nevertheless, even among livestock owners, we found a relationship between tolerance and direct experiences with carnivores. Livestock owners who had experienced damage from carnivore attacks were more inclined to have a negative attitude than those that suffered little or no damage. Hunters' attitude shifted from neutral to negative as they perceived the return of carnivores as a potential threat for hunting dogs. Moreover, they may see predators as competitors for large game species [54]. Of particular interest are the results obtained in terms of attitude comparisons between periods as far as conservationists are concerned. Contrary to the first period in which only positive attitudes were reported, during the second one, one negative [67] and two neutral attitudes [22,59] were registered. Niedziałkowski and Putkowska-Smoter (2020) [67] stated that some foresters benefitted from organising wolf hunts for Polish and international hunters, while others believe in and recognize the ecological value of wolves within the ecosystem (i.e., through limiting ungulate densities, they indirectly have a positive impact on forest plantations). However, at the local and regional levels, foresters did not particularly endorse wolf protection and sometimes outright criticized conservation initiatives. Anthony and Tarr (2019) [59] declared that some park members perceived the presence of wolves as positive since they reduce the number of damaging species such as wild boar, beyond removing weak and/or sick animals. On the other hand, others believe that all wolves should be killed or confined to zoos. Gosling et al. (2019) [22] found that foresters exhibited a neutral position, but such results may have been influenced by the fact that 35% of them were hunters and 32% held livestock. Despite a very small data set, these findings are interesting as they may suggest that, in line with carnivore recolonization, conservationists become aware that coexistence between people and predators may be impractical in some areas. Thus, they have probably started to change their exclusively conservationist position towards a conservationist and management one. However, as stated before, because of the small sample size, further research should be implemented to provide stronger inferences.

Because of the poor quality of the information available, comparisons in terms of attitudes between areas where humans and carnivores have always coexisted with those in which carnivores have been eradicated were carried out only with reference to rural inhabitants and hunters. In both cases, we observed a negative attitude in both areas, which was even more negative in recolonization zones. This result suggested that where carnivores and humans coexist, conflict occurs because of direct or indirect interactions. Thus, the attitude is generally negative. Moreover, in areas where people are no longer accustomed to coexisting with predators, they may perceive the return of carnivores as a sort of limit for either freedom [89] and/or livestock activities [88]. However, because of the very small number of studies reported, this result should be interpreted cautiously, and further research should be carried out.

From our research, we were unable to provide a broader evaluation in terms of degree of tolerance towards bears and wolves, as few studies (n = 3) reported discriminatory attitudes between the two species. Pohja-Mykrä (2016) [47] showed that hunters and livestock owners considered the wolf as the most problematic species. The same results were reported by Pohja-Mykrä (2017) [55] and Mykrä et al. (2017) [54], i.e., hunters and rural inhabitants showed a less positive attitude towards wolves than that shown towards bears.

3.5 Research Limits

We are aware that our research presents some limitations. Firstly, we focused only on peer-reviewed English literature, thus excluding the grey literature, which might have provided additional information. We decided to focus only on peer-reviewed studies because the scientific contribution represents an important focus of this work. Despite this, we believe that because of the nature of the topic, this exclusion criterion did not eliminate a large number of highly valuable studies. However, we recognize that the grey literature may represent a valuable source of information when more valuable peer-reviewed research is limited. Secondly, the attitude index value of each category involved was strongly affected by the number of studies in which such attitudes were obtained. Finally, marked differences emerged in terms of the number of studies published between the first (n = 13) and second period (n = 27) and between coexistence (n = 4) and non-coexistence areas (n = 16), which may have affected the results obtained.

3.6 Conclusions

The return of large carnivores in areas in which they were previously extirpated evoked different feelings among the different involved stakeholder categories. This not only refers to the European context but even includes other countries (e.g., India) in which human density has reached remarkable levels. Consequently, in such areas, interactions or conflicts between wild species (especially carnivores) and human activities may become very intense. Therefore, the synergistic participation of research institutions and local authorities should be implemented to find the most adequate solutions aimed at promoting coexistence in the long-term. Our findings support the idea that the return of both bears and wolves is challenging for conservation, as interactions with such predators may alter people attitudes, particularly those of the most affected categories. Long-lasting coexistence does not necessarily imply that people are more willing to accept carnivores. Conservationists should thus continuously monitor public attitudes, as opinions change when carnivore populations become established. Moreover, changes in these carnivores' demographic parameters need to be carefully taken into consideration by carnivore-policy makers to draw up effective management actions with the aim of minimising livestock killings. Because of the existence of a common EU policy, transboundary cooperation may help in the design of shared and effective mitigation strategies. In the near future, research institutions and management authorities will have to cooperate to solve a growing variety of human–carnivore conflicts, especially in those contexts in which political, social, and ecological conditions are changing. Therefore, communicating effectively with the public to promote large carnivore conservation and the maintenance of husbandry practices takes on a notable importance. We recognize the utility of extensive grazing practices in terms of the ecosystem services provided, especially at an EU level, and we are aware that if biodiversity eradication to promote the expansion of rural activities is impractical and dangerous, on the other hand, conservationists should change their views regarding the separation between rural practices and nature, throughout accepting that livestock practices are part of the ecosystem. Therefore, coexistence should be promoted and communicated, not only focusing on the importance of wildlife preservation, but also by highlighting the needs of livestock grazing practices of great economic, traditional, and ecological values.

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4. Crested porcupine (*Hystrix cristata*) abundance estimation using Bayesian methods: first data from a highly agricultural environment in central Italy

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4.1 Introduction

Estimating the abundance of wild species is a key issue in animal ecology. Because humans cannot constantly observe and/or monitor animals, assessing wildlife abundance is difficult and the field methods used to achieve this goal may be prone to bias (Iijima 2020). Block count and aerial surveys are frequently used to assess the abundance of wild species (e.g., Mourão et al. 1994; Salahudeen et al. 2013). Nevertheless, it is conceivable that data obtained from direct counts may contain notable variation depending on population dynamic and distribution, in turn, linked to the biological and ecological features of the target species (e.g., reproduction, migration) (Iijima 2020). Bayesian analyses are frequently used to overcome the potential imprecisions derived by field-work data collection (Iijima 2020). Modeling using Bayesian statistical inferences can increase the precision of model parameter because of the combination of previous knowledge (known as priors) with data collected in the field to produce *a-posteriori* distribution (McCarthy and Masters 2005; Martin et al. 2013; Morris et al. 2015). Capture-recapture is one of the most common methods used to assess animal abundance and it is frequently implemented with Bayesian analyses (i.e., spatially explicit capture-recapture models) (Efford 2004; Royle et al. 2014) to estimate the number of individuals inhabiting an area (Royle et al. 2017; Romairone et al. 2018). Although capture-recapture methods were originally developed to mark individuals, this approach can be extended without the need to physically capture animals (Royle et al. 2017). For example, throughout the use of camera traps, individual-based recognition can be done for those species presenting specific phenotypic traits in the form of stripes or spots (e.g., tigers, leopards) (Royle et al. 2017). Nevertheless, for species in which such traits are absent or hardly recognizable, individual identification is impractical and potentially prone to considerable biases. In these cases, assessing species abundance at a site without individual recognition could be less reliable and requires the use of alternative analyses. These so-called presence-only data are often collected during opportunistic surveys and are generally stored in online data bases and/or museums (Dorazio 2014). These data can be gathered using different opportunistic (e.g., road-killing events, citizen-science platforms) and/or systematic monitoring methods (e.g., radio-telemetry, camera traps) (Abadi et al. 2012; Wilson et al. 2016). However, the main limit is that they do not take into account for the errors in the detection of individuals (Chen et al. 2013). In spite of this limitation, analyses of presence-only data are partially motivated by the difficulties and expenses of conducting planned surveys of wild populations (Dorazio 2014) and could be used to provide a rough abundance estimate of those species, like crested porcupines *Hystrix cristata*, in which the absence of specific phenotypic markers do not allow for an accurate individual-based recognition.

The crested porcupine in Italy is strictly protected under the National Law 157/1992, whereas at the European level it is included within the Annex II of the Bern Convention (1979) and the Annex IV of the "Habitat" Directive 1992/43/ EEC. Before 1970, in the Italian Peninsula, it was present only in the central-southern regions facing the Tyrrhenian coast, and in the southern areas facing the Adriatic Sea (Toschi 1965). In Italy, the species is well and continuously distributed in the central regions (including Tuscany), while both north and southward presents a much more fragmented distribution (Mori et al. 2021). Nonetheless, to the best of our knowledge, to date information regarding its abundance at both small and large scale are still widely lacking. Factors including the abandonment of rural areas favored the species expansion because of the increase of woodland habitats, particularly suitable for porcupines especially for food and denning (Monetti et al. 2005; Lovari et al. 2013; Mori et al. 2014a, 2017; Mori and Fattorini 2019). In addition, although agricultural habitats are generally avoided, extensive cultivations may provide important food resources (especially during the warmer months) (Lovari et al. 2013; Mori et al. 2014a), as long as they are intermixed with patches of natural or semi-natural vegetated areas (Torretta et al. 2021).

As a consequence, although the negligible crop damages registered in central Italy compared to other species, for a total amount of 19,500 euros/year claimed in southern Tuscany in recent years (Lovari et al. 2017), the crested porcupine is widely precepted as one of the main agricultural pests and is frequently subjected to poaching (Laurenzi et al. 2016; Cerri et al. 2017; Lovari et al. 2017). Therefore, assessing the species abundance (especially in highly human-altered habitats) assumes remarkable importance to delineate adequate management and conservation strategies.

Using presence-only data obtained from camera traps, opportunistic observations, and road-killing events, here we present the first attempt to provide a rough estimate of the crested porcupine abundance within a highly agricultural environment in central Italy.

4.2 Materials and methods

4.2.1 Study area and data collection

The study was carried out in the Tuscany region (central Italy), within an area of approximately 18,073 km^2. Such an area was defined through a 100% Minimum Convex Polygon (100% MCP) and starting from the coordinates identifying camera trap locations and independent opportunistic observations of porcupines (Fig. 1).

For what concerns camera-trapping, we used independent presence-only data collected from previous studies realized in the whole Tuscany region during the year 2013. Because in these studies cameras were placed to detect multiple species and researches were not tailored to specifically address porcupine abundance (data available on www.inaturalist.org, and published works: e.g., Mori et al. 2014b; Franchini et al. 2017; Mori and Menchetti 2019; Viviano et al. 2021), there is a mismatch in terms of number of cameras placed in each province. Overall, we used information obtained from 134 cameras located in nine provinces: Arezzo (n = 19), Florence (n = 19), Grosseto (n = 67), Livorno (n = 3), Lucca (n = 6), Massa Carrara (n = 1), Pisa (n = 4), Pistoia (n = 1), and Siena (n = 14). Because cameras were distributed across the whole region (Fig. 1), the average distance between them was of about 74,749 m (Standard Deviation

(SD) = 41,887 m). Cameras were placed along trails and/or in the near proximity of denning sites at a height of 30–50 cm above the ground, activated 24 h per day, and checked every 15 days to download data and check for their functionality. The overall monitoring effort was of 4319 camera-trap/night recording an average number of 38 (SD = 22) porcupine independent records per camera. To be defined as an independent event, images of porcupines were filtered considering a time span of 30 min. between pictures at the same site (Meek et al. 2014).

Data on independent opportunistic observations (reported in 2013) were obtained from both iNaturalist (https://www. inaturalist.org/) and Ornitho (https://www.ornitho.it/) platforms, and from direct sightings reported by local people. Overall, 24 observations were reported in six provinces: Arezzo (n = 1), Grosseto (n = 18), Pisa (n = 1), Pistoia (n = 1), Prato (n = 1), and Siena (n = 2). Individuals identified through camera traps and/or independent opportunistic observations were all classified as adults or sub-adults based on body size (Mori *personal communications*).

Carcasses of 27 road-kill individuals (n = 8 males, n = 12 females, and n = 7 sex-unidentified individuals) were opportunistically collected in two provinces: Grosseto (n = 18) and Siena (n = 9) during the year 2013 (Fig. 1). Individuals were classified as adults (n = 13), sub-adults (n = 13), and young (n = 1) based on body measurements and tooth eruption (Mori and Lovari 2014).

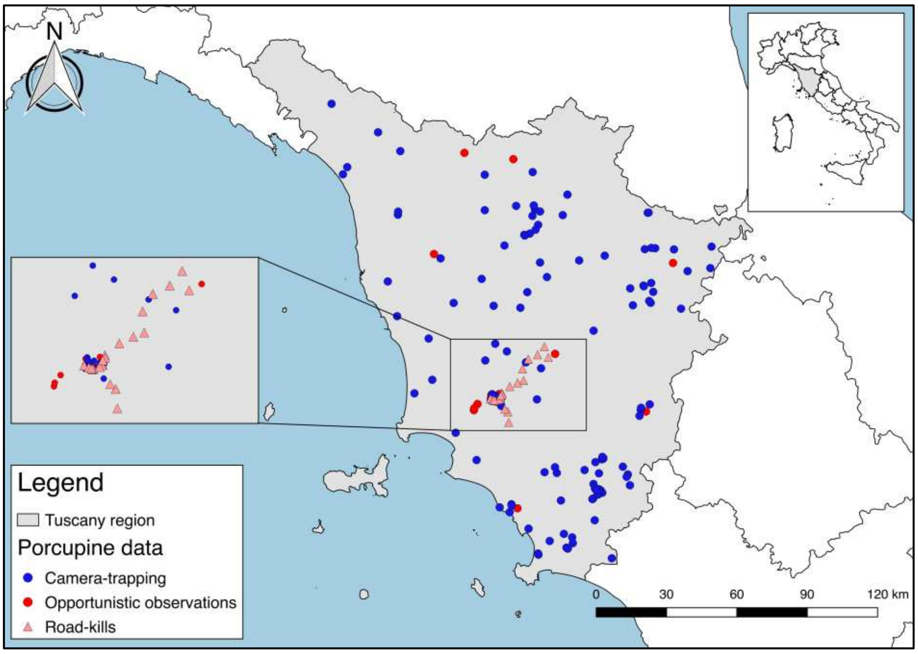

Fig. 1 Location of the study area (Tuscany region) with the relative camera trap (blue dots), independent opportunistic observation (red dots), and road-killing (pink triangles) coordinates. Road-kill individuals were collected in the only provinces of Grosseto and Siena

4.2.2 Spatial analyses

Spatial analyses were conducted using the QGIS Software (v. 3.18). Two buffers of 180 and 4896 m, respectively, were applied to each coordinate representing independent opportunistic observations and camera trap locations. The decision to apply such buffer sizes was taken considering the minimal and maximal dispersal distance known for both adult and sub-adult crested porcupines in central Italy (Mori and Fattorini 2019). Within each buffer, we counted the number of points representing either independent camera trap positive detections and/or independent opportunistic observations. Those buffers including only one point were considered as representative of only one individual roaming in the buffer area. On the contrary, more points falling within the same buffer were considered as spatially autocorrelated. Therefore, to avoid potential pseudo-replication biases, only one individual was assumed to roam within the buffer area.

For what concerns the habitat composition analyses, we used the 100% MCP calculated for each area (see "Statistical analyses" section and Fig. 2 for details) and we extracted the percentage of each land cover class starting from the Corine Land Cover 2012 (CLC) (EEA 2018) of the Tuscany region (excluding all the islands) and represented through a 100-m raster layer. We then re-classified the original CLC categories into five macro-habitat categories: agricultural areas, urban areas, canopy-covered and open areas (i.e., woodlands, forests, shrublands, grass- lands), water bodies, and cliffs, glaciers, and coastal areas (Table 1). Canopy-covered and open areas were considered as "suitable habitats," agricultural areas as "moderately suitable habitats," while urban areas, water bodies, and cliffs, glaciers, and coastal areas as "unsuitable habitats" (Monetti et al. 2005; Lovari et al. 2013; Mori et al. 2014a, 2017; Mori and Fattorini 2019; Torretta et al. 2021).

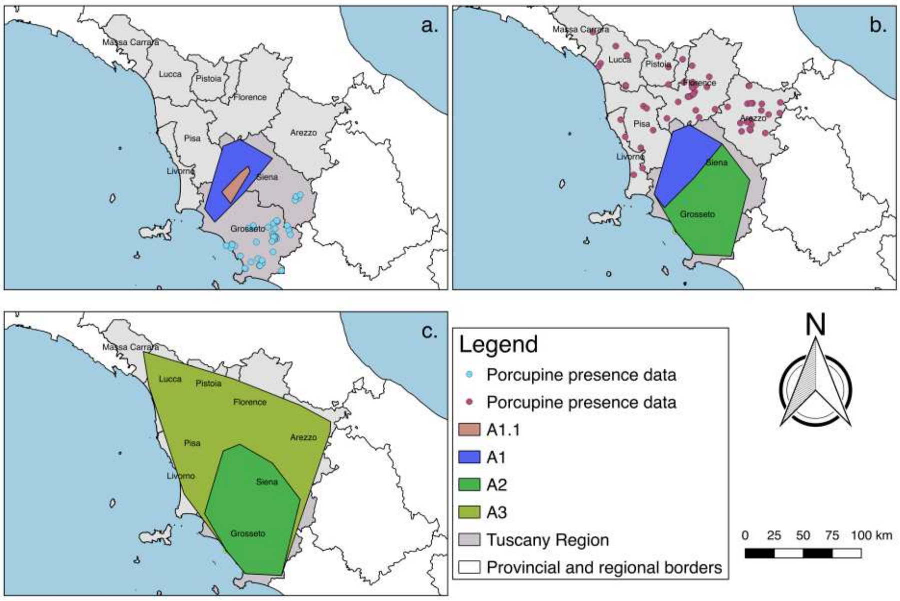

Fig. 2 (a) The 100% Minimum Convex Polygons (100% MCPs) applied to coordinates representing camera trap locations and independent opportunistic observations in the northern area of the provinces of Grosseto and Siena (A1 = 2044 km^2) and road-killing sites (A1.1 = 278 km^2). **(b)** The 100% MCP size comparison between A1 and the area covering the whole provinces of Grosseto and Siena (A2 = 6376 km^2). (c) The 100% MCP size comparison between A2 and the whole study area (A3 = 18,073 km^2). Porcupine presence data indicate coordinates of camera trap locations and independent opportunistic observations in the southern area of the provinces of Grosseto and Siena (light blue dots) and in the central/northern area of the whole study area (purple dots)

Table 1 Percentage of habitat composition obtained from the Corine Land Cover 2012 (CLC) map (EEA 2018) in the area defined by road-killing sites (A1.1), in the northern area of the provinces of Grosseto and Siena (A1), in the whole provinces of Grosseto and Siena (A2), and in the whole study area (A3) (see Fig. 2 for further details). Canopy-covered and open areas include woodlands, forests, shrublands, and grasslands

Habitat	A1.1 (278 km^2)	A1 (2044 km^2)	A2 (6376 km^2)	A3 (18,073 km^2)
Agricultural areas	32.38	45.45	57.06	50.31
Urban areas	1.47	2.3	1.69	4.61
Canopy-covered and open areas	66.15	52.21	40.75	44.37
Water bodies	0.00	0.01	0.15	0.47
Cliffs, glaciers, and coastal areas	0.00	0.03	0.35	0.24

4.2.3 Statistical analyses

To estimate the abundance of porcupines in each monitoring area (A1, A2, and A3 – see Online Resources, "Results" section, and Fig. 2 for details) we used a simple Bayesian model with just one parameter and considering only the surfaces covered by suitable and moderately suitable habitats for porcupines. The choice to divide the whole study area into three main areas was taken to best account for the spatial mismatch in terms of camera-trap locations and opportunistic observations, which may have strongly biased the calculation of the prior distribution and, consequently, the posterior one. Our count data, C, which indicate the likelihood, come from a Poisson distribution expressed through a parameter λ:

$C \sim$ Poisson (β)

λ represents the expected number of porcupines in the area surveyed and is calculated from the number of animals counted (or estimated) within each monitoring area, N, and the proportion of the area surveyed, a:

$\beta = N \times a$

The prior for N is represented by a Gamma distribution with parameters S (shape) and R (rate):

$N \sim$ Gamma (S R)

The Gamma distribution represents the conjugate prior distribution for a Poisson likelihood. Consequently, even the posterior distribution will be a Gamma distribution.

To obtain a better rough estimate of the number of porcupines inhabiting the suitable and moderately suitable habitats of the whole monitoring area (A3 – Fig. 2c), the analysis was divided into three steps (i.e., repeated for each of the three different monitoring areas) and using first a mathematical approach (to assess for the relationship among likelihood, prior, and posterior distributions), and then a computational one (to assess the goodness-of-fit of each model), through the implementation of the Software R (v. 4.0.1) (R Development Core Team 2021) in JAGS (Plummer 2003) using the "jagsUI" package (Kellner and Meredith 2021):

A1. *Mathematical approach*: we used a 100% MCP to calculate the area defined by each coordinate representing independent opportunistic observations and camera trap locations in the northern area falling within the provinces of Grosseto and Siena (A1 — Fig. 2a). Another 100% MCP was used to calculate the area defined by each coordinate representing the recovery sites of each road-kill porcupine (A1.1 — Fig. 2a) falling within the same larger area (i.e., A1 — Fig. 2a). λ was calculated starting from the number of road-kill porcupines multiplied by the ratio between A1 and A1.1 (which expresses the proportion of the area surveyed) and only considering the surface covered by suitable and moderately suitable habitats for the species. We then created a vector defining the interval input, i.e., the minimum and the maximum number of crested porcupines estimated within the northern area of the provinces of Grosseto and Siena (i.e., A1), obtained from the two buffers (i.e., 180 and 4896 m, respectively) applied to those coordinates representing both camera trap locations and independent opportunistic observations defining A1. The minimum value obtained from the interval was subtracted to λ, while the maximum value was added to the same. This was done in order to obtain a rough estimate of the range of individuals inhabiting A1. Since the number of individuals obtained from λ may be overestimated while the range of individuals obtained from the two buffers is most likely underestimated, using this approach we partially reduced such a bias. Thereafter, we calculated the mean and the standard deviation of the range. These data were used as input information to define both the shape and rate of the prior distribution. The combined information obtained from the likelihood and prior distribution were then used to produce the posterior distribution.

Computational approach: the model was run in JAGS using three chains, 20,000 iterations, zero burnin, and one thinning. The diagnostic plot used to assess the goodness-of-fit of the model was visualized using the *diagPlot* function, implemented in the "wiqid" package (Meredith 2017). The goodness-of-fit of the model was then evaluated considering the Rhat index, which represents the scale reduction factor indicating convergence between chains (Gelman et al. 2004; Gelman and Hill 2007; Kruschke 2014) and the MCEpc value, which expresses the error in the mean estimation (percentage value) compared to the target distribution (Lunn et al. 2013). Step-by-step calculations are provided in the Online Resource 1.

A2. *Mathematical approach*: we used a 100% MCP to calculate the area defined by each coordinate representing independent opportunistic observations and camera trap locations in the whole area including the provinces of Grosseto and Siena (A2 — Fig. 2b). The new λ was calculated starting from the average number of individuals estimated within A1 (posterior distribution obtained from the first-step analysis) multiplied by the ratio between A2 and A1 (proportion of area surveyed) and only considering the surface covered by suitable and moderately suitable habitats for the species. We then created a vector defining the range which, in turn, refers to the minimum and maximum number of crested porcupines estimated within the southern area of the provinces of Grosseto and Siena (Fig. 2a — light blue dots), obtained from the two buffers applied to those coordinates representing both camera trap locations and independent opportunistic observations. As done in step 1, the minimum value of the range was subtracted to the λ, while the maximum value was added to the same. Subsequently, we calculated the mean and the standard deviation of the range. These data were used as input information to define both the shape and rate of the prior distribution and implemented with the information obtained from the likelihood to obtain the posterior distribution.

Computational approach: the model was run in JAGS using three chains, 20,000 iterations, zero burn- in, and one thinning. The diagnostic plot was visualized using the *diagPlot* function, implemented in the "wiqid" package (Meredith 2017), and the goodness-of- fit of the model was assessed based on both the Rhat index (Gelman et al. 2004; Gelman and Hill 2007; Kruschke 2014) and the MCEpc value (Lunn et al. 2013). Step-by-step calculations are provided in the Online Resource 2.

A3. *Mathematical approach*: we used a 100% MCP to calculate the area defined by each coordinate representing independent opportunistic observations and camera trap locations in the whole monitoring area (A3 — Fig. 3c). As done in step 2, the new λ was calculated starting from the average number of individuals estimated within A2 (posterior distribution obtained from the second-step analysis) multiplied by the ratio between A3 and A2 (proportion of area surveyed) and only referring to the surface covered by suitable and moderately suitable habitats for the species. We then created a vector defining the range which, in turn, refers to the minimum and maximum number of crested porcupines estimated within the remaining monitoring area (Fig. 2b — purple dots), obtained from the two buffers applied to those coordinates representing both camera trap locations and independent opportunistic observations. The minimum value of the range was subtracted to λ, while the maximum value was added to the same. Afterward, we calculated the mean and the standard deviation of the range. These data were used as input information to define both the shape and rate of the prior distribution and implemented with the likelihood information to produce the posterior distribution.

Computational approach: the model was run in JAGS using three chains, 20,000 iterations, zero burnin, and one thinning. The diagnostic plot was visualized using the *diagPlot* function, implemented in the "wiqid" package (Meredith 2017), and the goodness-of- fit of the model was assessed based on both the Rhat index (Gelman et al. 2004; Gelman and Hill 2007; Kruschke 2014) and the MCEpc value (Lunn et al. 2013). Step-by-step calculations are provided in the Online Resource 3.

4.3 Results

The habitat analysis revealed that, in each area (A1.1, A1, A2, or A3), agriculture constituted much of the land covered, being preponderant in A2 and A3 while reaching the "second place" only in A1 and A1.1. Furthermore, in A1, such a value is comparable to the percentage of land covered by canopy-covered and open areas (Table 1).

Following the subdivision provided in the "*Statistical analyses*" section, results are presented in three steps:

A1. Within A1.1 (278 km^2) we opportunistically collected 27 carcasses of road-kill porcupines. Starting from the mathematical product between these 27 animals and the ratio between the area covered by suitable and moderately suitable habitats found in A1 (1996.17 km^2) and A1.1 (273.97 km^2) (proportion of area surveyed), we obtained a λ corresponding to 197 individuals. From the application of the two buffers (180 and 4896 m, respectively) to each coordinate representing independent opportunistic observations and camera trap locations falling within A1, we obtained a range spanning from 17 to 40 individuals. By subtracting the lower value (i.e., 17) to λ and adding the higher one (i.e., 40) to the same, we obtained an interval ranging from 180 to 237 individuals. The model output obtained from both the mathematical and computational approach (showing the posterior distribution) allowed us to estimate an average minimum number of 207 porcupines (SD = 15.41, 95% confidence interval (95% CI) = 176–237) within the suitable and moderately suitable habitats in A1 (Fig. 3) (see the Online Resource 1 for details).

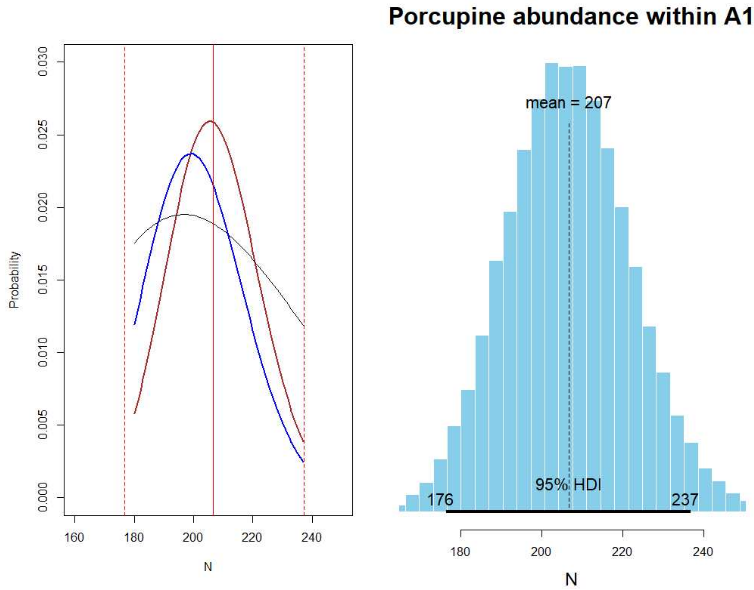

Fig. 3 The line chart on the left shows the trends of the likelihood (black line) and both the prior and posterior distributions (blue and brown lines, respectively). The bar plot on the right (posterior distribution) shows the average minimum number of porcupines estimated within the suitable and moderately suitable habitats falling in the northern territory of the provinces of Grosseto and Siena (A1 = 1996.17 km^2)

A2. Through the multiplication of the average minimum number of porcupines estimated during the first-step analysis (i.e., 207 individuals) and the ratio between the area covered by suitable and moderately suitable habitats found in A2 (6236.37 km^2) and A1 (proportion of area surveyed), we obtained a new λ corresponding to 647 individuals. From the application of the two buffers to each coordinate representing independent opportunistic observations and camera trap locations falling within the southern area of A2 (see Fig. 2a light blue points), we obtained a range spanning from 29 to 50 individuals. By subtracting the lower value (i.e., 29) to the λ and adding the higher one (i.e., 50) to the same, we obtained an interval ranging from 618 to 697 individuals. The model output obtained from both the mathematical and computational approach (showing the posterior distribution) allowed us to estimate an average minimum number of 655 porcupines (SD = 20.56, CI 95% = 614–695) within the suitable and moderately suitable habitats in A2 (Fig. 4) (see the Online Resource 2 for details).

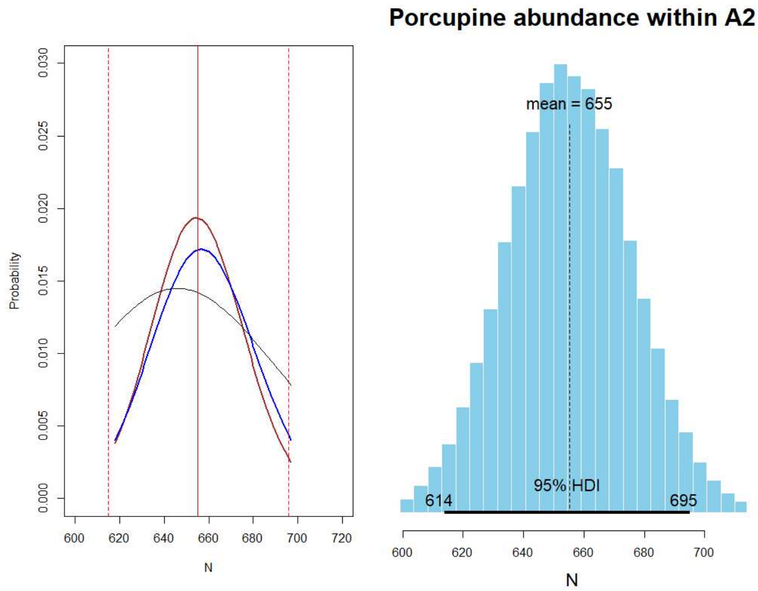

Fig. 4 The line chart on the left shows the trends of the likelihood (black line) and both the prior and posterior distributions (blue and brown lines, respectively). The bar plot on the right (posterior distribution) shows the average minimum number of porcupines estimated within the suitable and moderately suitable habitats falling in the whole area including the provinces of Grosseto and Siena (A2 = 6236.37 km^2)

A3. Through the multiplication of the average minimum number of porcupines estimated during the second-step analysis (i.e., 655 individuals) and the ratio between the area covered by suitable and moderately suitable habitats found in A3 (17,111.52 km^2) and A2 (proportion of area surveyed), we obtained a new λ corresponding to 1797 individuals. From the application of the two buffers to each coordinate representing independent opportunistic observations and camera trap locations falling within the central and northern area of A3 (see Fig. 2b purple points), we obtained a range spanning from 42 to 57 individuals. By subtracting the lower value (i.e., 42) to λ and adding the higher one (i.e., 57) to the same, we obtained an interval ranging from 1755 to 1854 individuals.

The model output obtained from both the mathematical and computational approach (showing the posterior distribution) allowed us to estimate an average minimum number of 1803 porcupines (SD = 26.89, 95% CI = 1750–1855) within the suitable and moderately suitable habitats in A3 (Fig. 5) (see the Online Resource 3 for details).

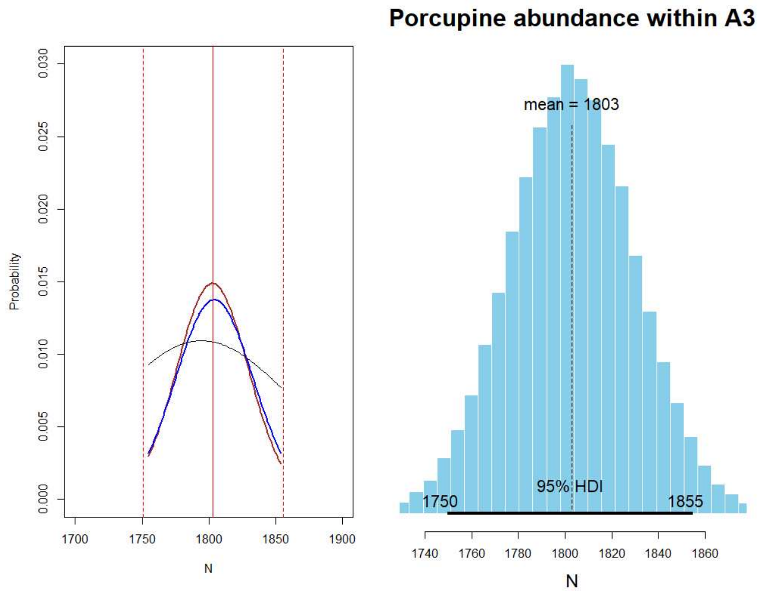

Fig. 5 The line chart on the left shows the trends of the likelihood (black line) and both the prior and posterior distributions (blue and brown lines, respectively). The bar plot on the right (posterior distribution) shows the average minimum number of porcupines estimated within the suitable and moderately suitable habitats falling in the whole study area (A3 = 17,111.52 km^2)

4.4 Discussion

To date, the crested porcupine is classified as “Least Concern” by the International Union for Conservation of Nature (IUCN) (Amori and De Smet 2016), despite being almost rare in Central African countries (Viviano et al. 2020). Nevertheless, in spite of this classification, data referring to its population trend are still widely lacking (Amori and De Smet 2016). Following the Resource Dispersion Hypothesis (RDH) (Macdonald 1983; Carr and Macdonald 1986), the distribution and quality of resources (e.g., food, shelters, partners, and/or sites for reproduction) affect species presence and distribution at a site. In fact, individuals are more inclined to occupy the smallest areas which contain all the resources they need (Harestad and Bunnel 1979). In the case of the crested porcupine, as reported by Lovari et al. (2013), the home-range size of a male may vary from 10.0 to 398.7 ha while that one of a female range from 18 to 478.15 ha. This means that, on average, the home-range of an individual (male or female) is of about 226.21 ha. Following these data and assuming all individuals as territorials, if we consider the surface covered by suitable and moderately suitable habitats in each area, A1 would be expected to host at least 883 individuals, while A2 and A3 would be expected to host 2759 and 7571 individuals, respectively, with an average minimal density of about 0.44 ind./100 ha for each area. Nevertheless, we believe that our estimates (0.10 ind./100 ha per area) are most closely related to the true number of individuals because of four reasons: (i) both intra- and inter- specific competition, along with the low reproductive rate of the species (from one to three births per pair per year, each composed by on average one or two porcupettes) (Coppola and Felicioli 2021), may play a key role in shaping species distribution and abundance at a site; (ii) road-killing and poaching events may substantially affect the species' survival capacity; (iii) our analysis included also sub-adult individuals which notoriously disperse before settling and, hence, do not show a territorial behavior (Mori and Fattorini 2019); and (iv) each area is covered by a considerable percentage of agriculture. Therefore, because homogeneous agricultural areas are considered as non-optimal for the species (Torretta et al. 2021), we believe that the carrying capacity of each area is lower than expected. However, in spite of these considerations, extensive cultivated fields may become hospitable for the species as long as they are intermixed with either natural or semi-natural vegetated areas, which in turn provide shelters and abundant food resources (Lovari et al. 2013; Mori et al. 2014a; Torretta et al. 2021). The porcupine is considered as a "potentially problematic species" because of damages to croplands, riverbanks, and tree debarking (Laurenzi et al. 2016; Lovari and Riga 2016; Lovari et al. 2017) and is subjected to persecution by local farmers (Cerri et al. 2017). Seeds, fruits, epigeal parts, roots, and other underground vegetables constitute the staple of the diet of the porcupine in Italy (Zavalloni and Castellucci 1994; Bruno and Riccardi 1995). However, corn, potatoes, pumpkins, sunflowers, and melons are consumed if locally available (Mori et al.

2013; Bertolino et al. 2015), thus reducing the tolerance of local farmers (Cerri et al. 2017). Our data suggest that the density of the species is still relatively low within the monitoring areas. However, the information obtained may help managers and conservationists to delineate appropriate actions aimed to reduce the potential negative impacts of porcupines over human activities, through the implementation of mitigation measures especially in those areas characterized by extensively cultivated fields and where the stable presence of the species may lead to agricultural damages.

Our research represents the first important contribution in the assessment of the crested porcupine abundance in a highly agricultural environment of central Italy. Furthermore, the Bayesian method we proposed in conjunction with spatial analyses (implemented to define the minimum and maximum number of individuals roaming within each buffer area), to our knowledge, represents the first attempt to estimate the abundance of a species using presence-only data (i.e., Bayesian model with just one parameter). Specifically, it is advantageous being of relatively simple implementation thus allowing to obtain a rough estimate of the target species abundance within an area, even in the absence of presence/absence data. Nevertheless, despite the results presented, we are aware that our study presents some limitations: (i) as stated above to estimate porcupine abundance we used presence-only data which, compared to presence-absence data, does not account for the imperfect detection (i.e., detection probability) (Chen et al. 2013) and/or the recovery probability (in the case of road-kill individuals); (ii) independent opportunistic observations were collected from citizen-science platforms which, despite their usefulness in the analysis of ecological data has already been assessed (e.g., Franchini et al. 2021), present several limitations (e.g., absence of a survey protocol, unknown sampling effort, no standardization, and poor *a-priori* control over observer quality) (Kery and Royale 2020); (iii) the mismatch in the spatial extent of camera traps and independent opportunistic observations did not allow us to provide stronger ecological inferences, especially regarding the potential difference in terms of the species abundance at a small-scale level. Indeed, the most likely different availability of resources (not evaluated in this study) may lead porcupines to be mostly abundant in some habitats and to a lesser extent in others; (iv) without being supported by field-data collection and validation, the number of porcupines estimated within each area could be most likely underestimated and, therefore, needs to be considered as a minimum referring value. In this sense, further researches involving spatially homogeneous field-based monitoring activities among areas and in conjunction with more appropriate statistical methods that take into account imperfect detection, are strongly suggested to provide further and detailed inferences regarding the abundance of the species in central Italy.

4.5 References

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5. Hotspot of human–carnivore conflicts in the north–eastern Italian Alps: conservation and management implications

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5.1 Introduction

Assessing the intensity of human–carnivore negative interactions is amongst the most challenging issues that conservationists and carnivore–policy makers face today. In fact, despite decades of research activities and resources invested on this topic, the main ecological and social conditions mostly affecting these interactions are still partially unknown (Dickman 2010). The negative human–carnivore interactions mostly refer to livestock predations and can be broadly attributed to several factors including large home–ranges of carnivores, their feeding behaviour, management interventions, and habitat loss and/or fragmentation (Broekhuis *et al*. 2017; Wilkinson *et al*. 2020). In fact, despite large carnivores typically do not permanently occur in high human–densely populated areas, they have shown a capability to re–colonize areas with moderate human densities, and to persist in fragmented lands (e.g., forests intermixed with agriculture) and/or in the near proximity of human buildings (Chapron *et al*. 2014) Therefore, in these areas, conflicts with humans are potentially more prone to occur. Carnivore attacks on livestock have become a major threat for carnivore conservation and human well–being, since carnivores may imperil the economic security of people mostly relying on grazing practices in turn leading to retaliatory killings (Miller 2015; Naha *et al*. 2018; Morehouse *et al*. 2020). Large carnivores are indeed amongst the most challenging species to preserve in our modern world. In fact, there is a deeply rooted hostility towards these species because of the real impact and/or perceptions of their negative impacts on human livelihoods (Dressel *et al*. 2015; Franchini *et al*. 2021).

In the Italian Alps, livestock activities provide important ecosystem services and represent an important part of the traditional and cultural heritage (Franchini *et al*. 2021). In this environmental context, the two most abundant large carnivores which may come into conflict with human activities are the grey wolf *Canis lupus* (hereafter, wolf) and the brown bear *Ursus arctos* (hereafter, bear). As for the wolf, the species has been extirpated along the Alps in the first twenty years of the 20th century and for decades was confined in the southern area of the Po River, with a small population composed of about 100 individuals survived in the central Apennines (Fabbri *et al*. 2007). From the 1970s onwards, a progressive and slow recovery occurred favoured by several ecological and social factors. Among these, we have the legal protection of the species occurred for the first time in 1971 (and later strengthened through a Ministerial Decree in 1976), and the abandonment of rural practices (especially in mountainous territories) which encouraged a rewilding process and the return of the wolf's natural prey (Fabbri *et al*. 2007; Boitani & Salvatori 2017). Thanks to the ecological connectivity between the north–western Apennines and the Ligurian Alps, the wolf re–appeared in western Alps in the 90s (Marucco *et al*. 2018). To date, the species is widespread in both the western and eastern Alps (Marucco *et al*. 2022) with a minimum estimated number of 946 individuals (range: 822 – 1099) (Marucco *et al*. 2022).

As for the bear, only one viable population is present along the central Italian Alps (AA.VV. 2010; Tosi *et al*. 2015). In the past, the species was widespread across the entire Alpine and pre–Alpine regions, until the Val Padana. However, it started to decline at the end of the 18th century due to logging activities (to create open fields to be dedicated to agriculture and livestock grazing practices) and hunting, which have led to its complete extinction in the eastern Alps (AA.VV. 2010; Tosi *et al*. 2015). Subsequently, between the first half of the 19th century and the 1930, the population gone also largely extinct in the central–western Alps, except for a small nucleus of individuals survived in the "Brenta" mountains (eastern Trentino) (AA.VV. 2010; Tosi *et al*. 2015). Nevertheless, even this small population has suffered a progressive numeric reduction up to its almost extinction at the end of the 90s (AA.VV. 2010; Tosi *et al*. 2015). In the attempt to save the Alpine bear population, between the 1999 and 2002, a European LIFE project called "LIFE *Ursus*" was carried out in collaboration between the Autonomous province of Trento and the

National Wildlife Institute (AA.VV. 2010; Tosi *et al*. 2015). The aim of the project was to restore a viable bear population of about 40–60 individuals in the middle/long term. To achieve this goal, ten individuals (seven females and three males) were translocated from the Slovenian population within the "Adamello Brenta" Natural Park (AA.VV. 2010; Tosi *et al*. 2015). To date, thanks to conservation efforts, the species counts more than 100 individuals mainly distributed in the Autonomous province of Trento and neighbouring areas (Groff *et al*. 2021). Another small nucleus of bears is present in the Friuli Venezia Giulia region (north–eastern Italy) which is composed by only dispersal males coming from either the Dinaric or central Italian Alps. Usually, in this region, from one to seven individuals are genetically identified each year (Franchini *et al*. 2022).

The return of these predators have led to an increasing number of negative interactions with human activities (e.g., Dondina *et al*. 2015; Tosi *et al*. 2015). To better counteract the effect of these (in some cases) severe damages, carnivore–policy makers have started to adopt different strategies spanning from the implementation of prevention measures (e.g., Mattiello *et al*. 2012; Berzi *et al*. 2021) to compensation payments (e.g., Boitani *et al*. 2010; Berzi *et al*. 2021; Galluzzi *et al*. 2021). This policy has been contributing to reduce both livestock and financial losses experienced by livestock owners. However, because the impact of carnivores on livestock may vary depending on several factors including environmental conditions (Broekhuis *et al*. 2017; Wilkinson *et al*. 2020), both livestock and wild prey availability (Khorozyan *et al*. 2015), and the feasibility of the implemented prevention measures which may vary depending on the context (Berzi *et al*. 2021), it becomes important to understand patterns of negative interactions to device effective mitigation strategies in those areas in which conflicts are more prone to occur (Ankit *et al*. 2021).

Statistical methods like hotspot analyses and/or predation risk modelling have started to be used as suitable methods to identify hot or coldspot conflictive areas, especially in recent years (e.g., Broekhuis *et al*. 2017; Ratnayeke *et al*. 2018; Hipólito *et al*. 2020; Ankit *et al*. 2021). In fact, for the effective conservation and management of potentially conflictive species like large carnivores, identifying hotspot areas of negative interactions assumes remarkable importance to identify those areas in which mitigation interventions should be prioritized (Miller 2015; Broekhuis *et al*. 2017; Hipólito *et al*. 2020; Ankit *et al*. 2021).

The first purpose of this work was thus to identify those areas in which conflicts between large carnivores (i.e., bear, wolf) and livestock practices were more intense (i.e., hotspot conflictive areas). Specifically, we asked the following research question: *Is there a spatial overlap between hotspot clusters of bear/wolf attacks and hotspot clusters of mountain farms/livestock abundance at a municipal level?* To answer this question, we formulated our hypothesis (H) as the 'species–specific foraging behaviour hypothesis'. H dictates that wolves are potentially more problematic than bears in terms of livestock attacks because of the different feeding behaviour of the two species. In fact, the former are more carnivorous (> 70% of the diet composed by animal component) (Zlatanova *et al*. 2014) while the latter, despite being classified among the large carnivores, are omnivorous, thus presenting a wider food spectrum and feeding largely on fruits and vegetables depending on the latitude and seasonal availability (Bojarska & Selva 2012). H predicts that we would expect to observe a spatial overlap between hotspot clusters of wolf attacks and hotspot clusters of mountain farms/livestock abundance at a municipal level. Conversely, we would not expect to observe the same relationship as far as bear attacks are concerned. Lastly, the second purpose of this work was to characterize bear and wolf attacks to discern the main factors which may increase the risk of attacks towards livestock. We assumed that the abundance of both bears and wolves would contribute to increase the number of attacks. Therefore, we would expect to observe higher livestock predations in those areas characterized by higher carnivores' abundance. We also have assumed that bear predations towards livestock would have linked to vegetation productivity recorded during years. We thus would expect to observe higher bear predations in those years characterized by lower vegetation productivity.

5.2 Materials and Methods

5.2.1 Study area

The study area was located in the central and north–eastern Italian Alps, including two administrative regions (Friuli Venezia Giulia, Veneto) and the Autonomous province of Trento (**Fig. 1**). The Friuli Venezia Giulia (hereafter, FVG) is the north–easternmost Italian region. The climate change in relation to the altitude: Mediterranean along the Adriatic coast, temperate–humid in both hilly and lowland areas, and Alpine in mountainous areas. The average annual temperature is of about 14.5 °C with abundant precipitation especially in the pre–Alps (up to 3,000 mm/year) and during autumn (~ 1,200–1,400 mm/year) (Franchini *et al*. 2019; Filacorda *et al*. 2021). The Veneto region is located between FVG and the Autonomous province of Trento. The climate changes from mild along the Adriatic coast and in hilly areas, continental on the plains, and Alpine along the mountains. The average annual temperature is of about 13.5 °C. Precipitations are scarce in winter (~ 750 mm/year) especially in lowlands. However, they became more abundant at higher altitudes (~ 750–1,100 mm/year) with the highest values recorded in the 'Bellunesi' pre–Alps (up to 3,200 mm/year). The Autonomous province of Trento is located in the central Italian Alps. The climate can be considered as transitional between sub–continental and Alpine. The average annual temperature is of about 12 °C with average annual precipitations ranging from 1,200–1,400 mm/year, and mostly abundant at higher elevations.

Habitat varies with location and altitude, with forests, shrublands and open habitats abundant in Alpine and pre–Alpine areas, while agriculture and urban areas mostly common in lowland territories. Extensive livestock practices are conducted in Alpine and pre–Alpine pastures and mainly during spring and summer, thus matching with the transhumant period (May–October) during which livestock is moved from lower to upper altitudes to graze in open areas and, thus, are more prone to be attacked by predators.

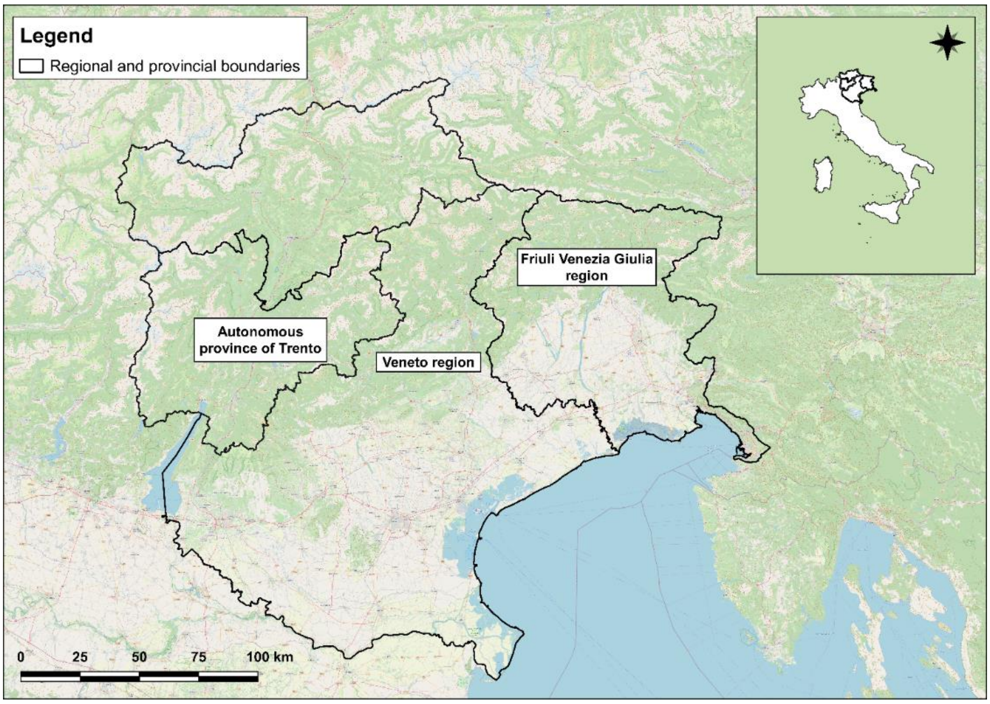


Figure 1. Location of the study areas (bold boundaries – inset map).

5.2.2 Data collection

To describe the impact of both bears and wolves on extensive grazing practices we analysed data of official claims collected from 2012 to 2020 by university researchers, wildlife technicians, veterinarians, and members of the forestry service working for each region/province. Requests for compensations for damages caused by large carnivores were presented within 24–48 hours (variable among regions and province) from the date of the statement. After an *in–situ* kill–site inspection conducted by trained personnel, both the cause of death as well as the eventual responsible predator were determined. Information regarding the date, municipality, location (coordinates in which the attack occurred), responsible predator (bear, wolf), attacked livestock species (cattle, sheep and goats), and number of predated/wounded individuals were collected and stored into a database.

5.2.3 Hotspot analysis

The hotspot analysis is a method of spatial analysis whose aim is to identify clusters of spatial phenomena (depicted in the form of points or polygons) on a map and refers to locations of events or objects (ESRI 2022a). This method has already been used to identify hotspots of human–carnivore conflicts in Asia (Ankit *et al*. 2021) and Europe (Hipólito *et al*. 2020) and relies on the *Getis–Ord Gi* statistics*, which aggregates points or polygon data into weighted features to find statistically significant spatial clusters of high or low values which in turn refer to hot or cold spots, respectively (Ankit *et al*. 2021; ESRI 2022a). The *Getis–Ord Gi* statistics* generates *p*–values (statistical probability) and *z*–scores (standard deviation) for each bin which indicate the comparison of significant clusters with neighbouring bins. A *z*-score higher than 1.96 or lower than -1.96 indicates the presence of a statistically significant hot or cold spots, respectively, compared to an average reference value (Ankit *et al*. 2021; ESRI 2022a).

Before implementing the hotspot analysis, we assessed both (*i*) the spatial autocorrelation and (*ii*) the incremental spatial autocorrelation of the dataset. Assessing the spatial autocorrelation allows to determine the presence of clustered or randomly distributed data. Clustered data distribution indicates the presence of spatial autocorrelation and, therefore, the presence of hot or coldspots (ESRI 2022b). The presence of one feature with high or low *z*–score value does not reflects the presence of a significant hot or coldspot. In fact, to be considered as a statistically significant spot, a feature must have a high/low *z*–score value and be surrounded by other features presenting high/low *z*–score values likewise (ESRI 2022a, b). The spatial autocorrelation was checked through the *Spatial Autocorrelation (Morans I)* function, implemented in the 'spatial statistics' toolbox, which lays on the Moran's Index value (ESRI 2022b). The incremental spatial autocorrelation measures spatial autocorrelation for a series of distances and, optionally, creates a line graph of those distances and their corresponding *z*–scores. The latter reflect the intensity of spatial clustering, and statistically significant peak *z*–scores indicate distances where spatial processes promoting clustering are most pronounced (ESRI 2022c). The incremental spatial autocorrelation was measured through the *Incremental Spatial Autocorrelation* function implemented in the 'spatial statistics' toolbox. As a starting distance, we used the average distance (m) among layers calculated through the *Calculate Distance Band* *from Neighbor Count* function, and considering an increment of 50 m. Thereafter, we realized the hotspot analysis using the *Hot Spot Analysis (Getis–Ord Gi*)* function. If the incremental spatial autocorrelation revealed the presence of peaks, the distance reported by the maximum peak was used as a threshold distance in the hot-spot analysis (ESRI 2022d). If no peaks were shown, we used the average distance calculated through the *Calculate Distance Band from Neighbor Count* function.

The hotspot analysis was realized at a polygon level, i.e., starting from the shapefile layer revealing the municipal boundaries. To assess for the variation in terms of clusters of attacks among years, the analysis was realized starting from the number of attacks registered in each municipality per year. Thereafter, we considered a cumulative trend of attacks from 2012 up to 2020 (i.e., 2012 vs 2012 + 2013 vs 2012 + 2013 + 2014, etc.) and divided by predator. This allowed us to observe both the change and intensity of predatory events over the years, also taking into consideration the amount of time in which predators are settled in the area. As for mountain farms, we started from the coordinates of each farm and then we implemented the hotspot analysis considering the number of farms falling within each municipality. The same was done for livestock, considering the number of cattle, sheep and goats reported for each municipality in the National Livestock Database (hereafter, BDN) (https://www.vetinfo.it/).

The hotspot analysis was realized using the ArcGIS Software (v. 10.5) (ESRI).

5.2.4 Statistical analysis

To characterize the impact of both wolves and bears on livestock practices we run Generalized Linear Mixed Models (GLMMs) (one per predator), using the 'lme4' R package (Bates *et al*. 2015), with residuals showing a negative binomial distribution. The family distribution of the response variable was assessed using the 'fitdistrplus' package (Delignette–Muller & Dutang 2015). As a response variable, we used the overall number of wounded/predated individuals per municipality and year. As a random factor we considered the municipality in which each predatory event occurred. This was done also to consider the spatial autocorrelation among data. As fixed factors we considered (*i*) the period in which each attack occurred, (*ii*) the year (2012–2020) in which each attack occurred, (*iii*) the average value of Normalized

Difference Vegetation Index (NDVI) per year (only for bears), which is considered as one of the best indicators of vegetation productivity (Wang *et al*. 2004) and was calculated using the MODIS NDVI Times Series Animation in google earth engine (https://earthengine.google.com/), (*iv*) the livestock density (n. ind./ha) (only referring to cattle and sheep and goats) at a municipal level, (*v*) the surface covered by each pasture (ha) in each municipality, (*vi*) the number of mountain farms in each municipality, (*vii*) the minimum abundance of each predator (bear, wolf) in each region and in the Autonomous province of Trento, and (*viii*) the attacked livestock species. The period was divided into transhumant (from May to October) and non–transhumant (from November to April). For both the FVG and Autonomous province of Trento, the surface covered by each pasture was calculated starting from the shapefile polygon layers provided by the administrative staff working for each institution. Conversely, for the Veneto region, we extracted and calculated the surface covered by each pasture starting from the Corine Land Cover (CLC) map of 2018 (Copernicus Land Monitoring Service – https://land.copernicus.eu/). The number of mountain farms in each municipality was calculated starting from the shapefile point layer provided by the administrative staff working for each institution. The minimum number of bears in FVG per year was defined based on the results obtained from the monitoring activities conducted by the researchers of the University of Udine (Vezzaro *et al*. 2018; Franchini *et al*. 2022). On the contrary, the minimum number of bears genetically identified in the Autonomous province of Trento per year was obtained considering the information reported in the provincial reports (Groff *et al*. 2013, 2014, 2015, 2016, 2017, 2018, 2019, 2020, 2021). The overall minimum number of bears genetically identified in FVG and the Autonomous province of Trento per year was reported in **Appendix 1**. The minimum number of wolves genetically identified in each region and in the Autonomous province of Trento per year was obtained from the LIFE WolfAlps reports (Marucco *et al*. 2018, 2022). Because the year 2019 was lacking those data, we considered the minimum number of wolves reported in 2020 within the LIFE WolfAlps report 2021/2022 (Marucco *et al*. 2022), by assuming that the number of wolves wouldn't have change consistently from 2019 to 2020. However, because the expansion of the wolf across the Alps is an ongoing dynamic process, we recognize the potential limits of this hypothesis. When the minimum number of wolves per region/province was not available and only information related to the minimum number of packs and/or couples were reported, we considered five individuals per pack which represents the average size of a wolf pack in the Alps (Grandi Carnivori in Trentino – https://grandicarnivori.provincia.tn.it/Il-lupo/BIOLOGIA-HABITAT-E-DISTRIBUZIONE/VITA-DI-BRANCO). The overall minimum number of wolves estimated and/or genetically identified in FVG, Veneto and the Autonomous province of Trento per year was reported in **Appendix 2**.

To encourage the maximum likelihood estimates of parameters, covariates were standardized using the *mutate_at* function implemented in the 'dplyr' R package (Wickham *et al*. 2022). To improve model convergence, we used an optimizer using the function *nlminb* implemented in the R package 'optimx' (Nash & Varadhan 2011; Nash 2014). Before being included in the model, the multicollinearity among independent variables was tested using the Variance Inflation Factor (VIF), implemented in the 'car' R package (Fox & Weisberg 2019), and considering VIF ≥ 5 as a threshold value to define highly correlated variables (Akinwande *et al*. 2015). Those variables showing a VIF ≥ 5 were considered as highly correlated and then the one showing the highest value was dropped from the model (Gareth *et al*. 2014; Peter & Bruce 2017). Model simplification was done using the *principle of parsimony* (Occam's razor) based on which a maximal model was fitted and then simplified through the removal of non–significant explanatory variables. Model ranking was done based on the Akaike's Information Criterion (AIC) (Akaike 1974) and ΔAIC (Burnham & Anderson 2002). In the presence of models showing ΔAIC < 2 (hence, considered as competitors of the best model) (Burnham & Anderson 2004) we performed model averaging by calculating the Akaike's weight (ωi), which express the relative amount of variation explained by each model compared with all other models (Burnham & Anderson 2002).

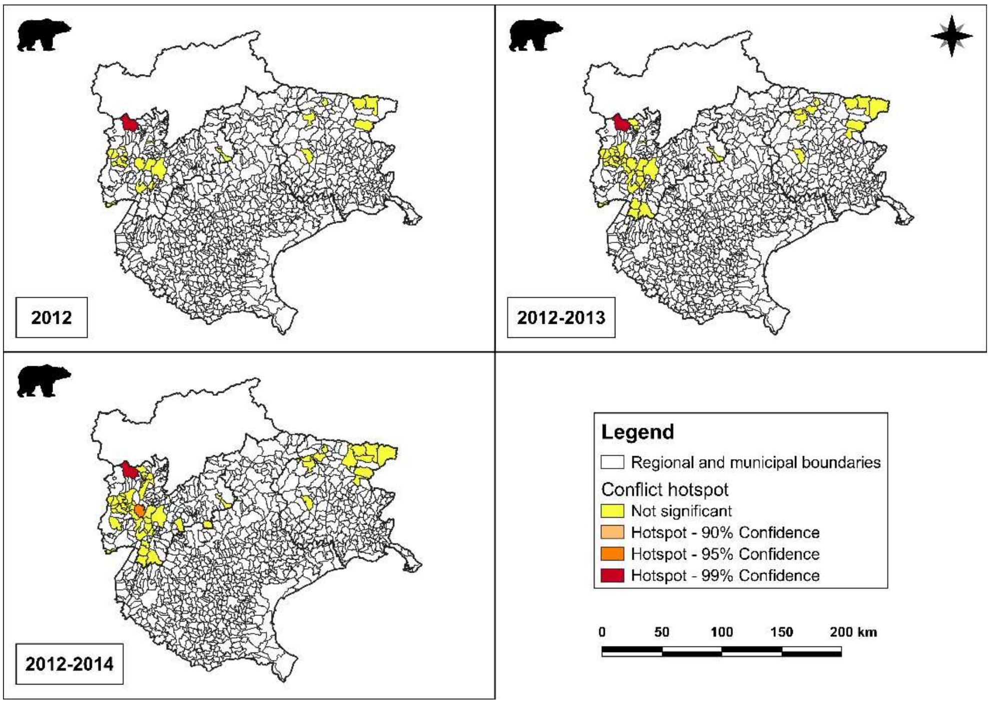
We explored the intensity of attacks among daily periods (i.e., early in the morning, late in the morning, day, late afternoon, evening, night), habitats in which they occurred (canopy–covered, fragmented pasture, open pasture, stable, other – ski run, hotel park, undefined areas), distance from the nearest human building, climatic conditions in which they occurred (clear weather, cloudy, foggy, snowy, raining/stormy), and cattle belonging to different age classes (< 6 months, 6–12 months, 12–18 months, 18–24 months, > 24 months) only considering wolf predations recorded in the Veneto region from 2018 to 2020. This choice was taken because the most detailed information were reported for that carnivore species, area and period. Because some values in the contingence table were < 5, the significance in terms number of predatory events among (*i*) daily periods, (*ii*) habitats, and (*iii*) climatic conditions was tested using the Fisher's test (Fisher 1922). Conversely, since all the values in the contingence table were > 5, the significance in terms of number of predatory events among (*i*) classes of distance from the nearest human building, and (*ii*) cattle belonging to different age classes was tested using the chi–squared test (Fisher 1922). The *pairwise nominal independent function (pnif)*, implemented in the R package 'rcompanion' (Mangiafico 2022), was thereafter implemented to assess the significance between pairs of groups.

The statistical analysis was conducted using the R Software (v. 4.2) (R Development Core Team 2022) and setting the *alpha* value (i.e., level of significance) at 0.05.

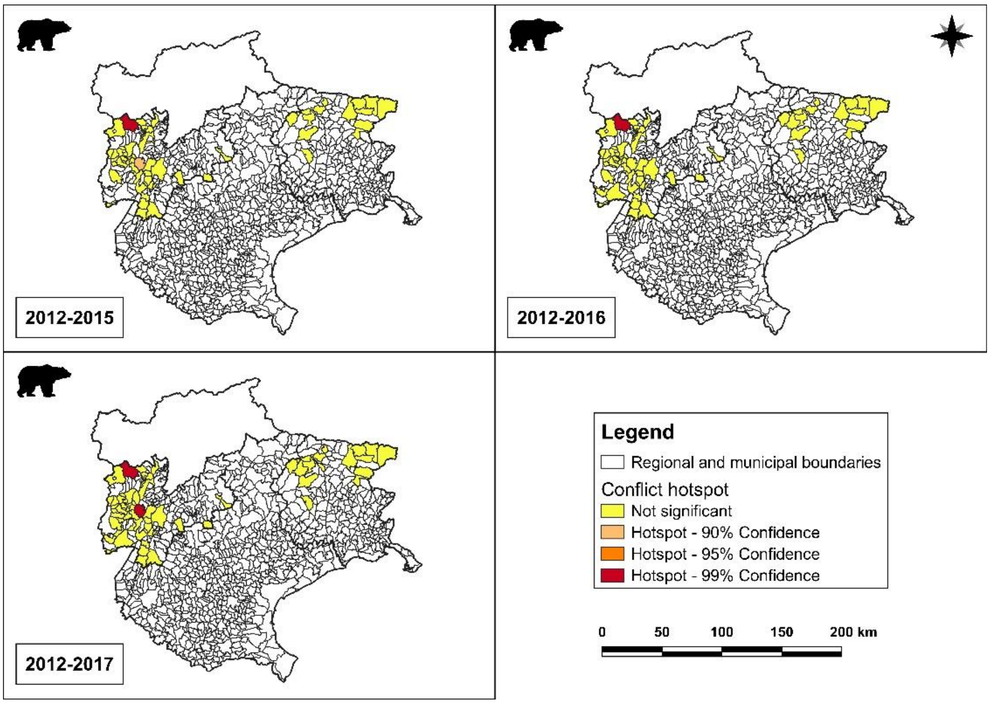
5.3 Results

5.3.1 Hotspot conflictive areas

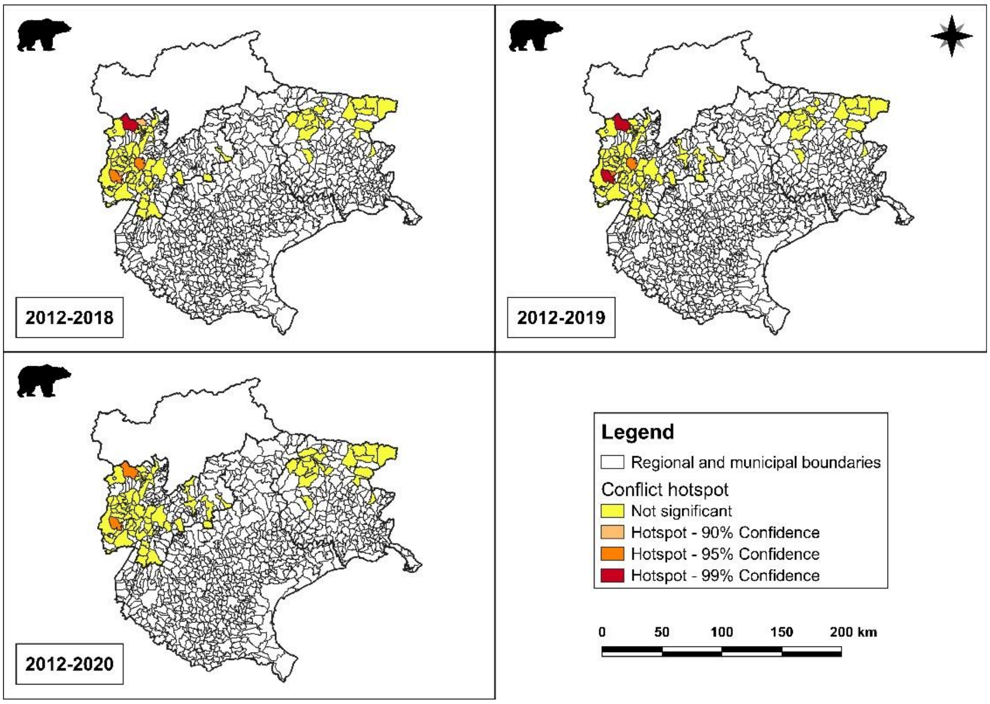
With regards to bears, no hotspot conflictive areas were observed in the considered time intervals, i.e., 2012–2014 (**Fig. 2a**), 2012–2017 (**Fig. 2b**), 2012–2020 (**Fig. 2c**), since from the spatial autocorrelation analysis it turned out that data were randomly distributed (**Appendix 3**).


2012
2012-2013
2012-2014
Legend
Regional and municipal boundaries
Conflict hotspot
Not significant
Hotspot - 90% Confidence
Hotspot - 95% Confidence
Hotspot - 99% Confidence
0 50 100 150 200 km

(a)


2012-2015
2012-2016
2012-2017
Legend
Regional and municipal boundaries
Conflict hotspot
Not significant
Hotspot - 90% Confidence
Hotspot - 95% Confidence
Hotspot - 99% Confidence
0 50 100 150 200 km

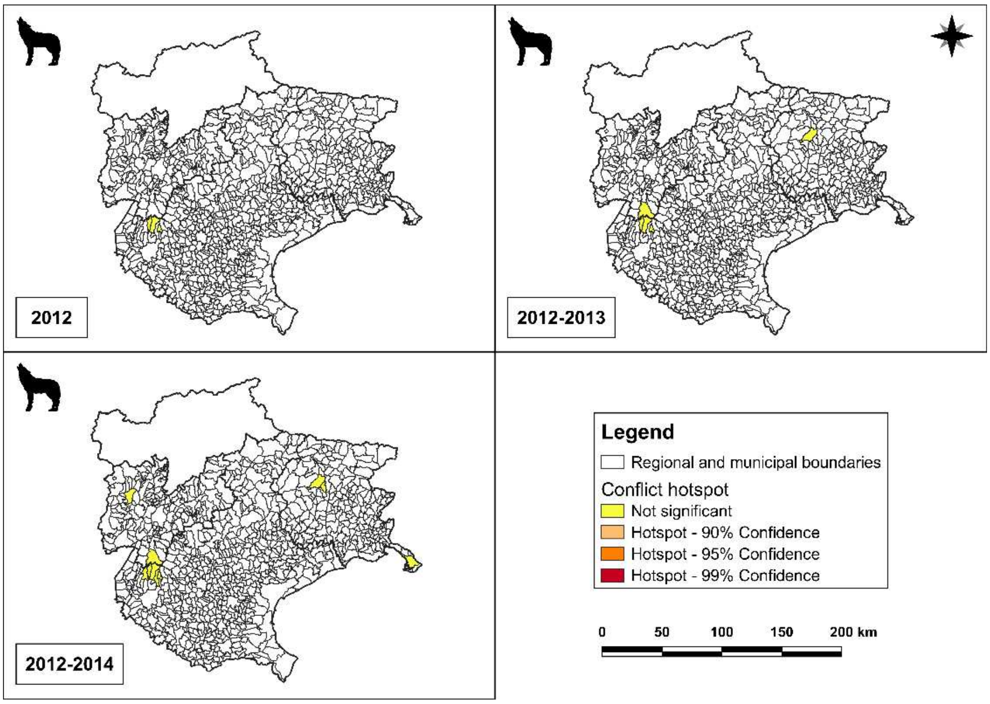
(b)



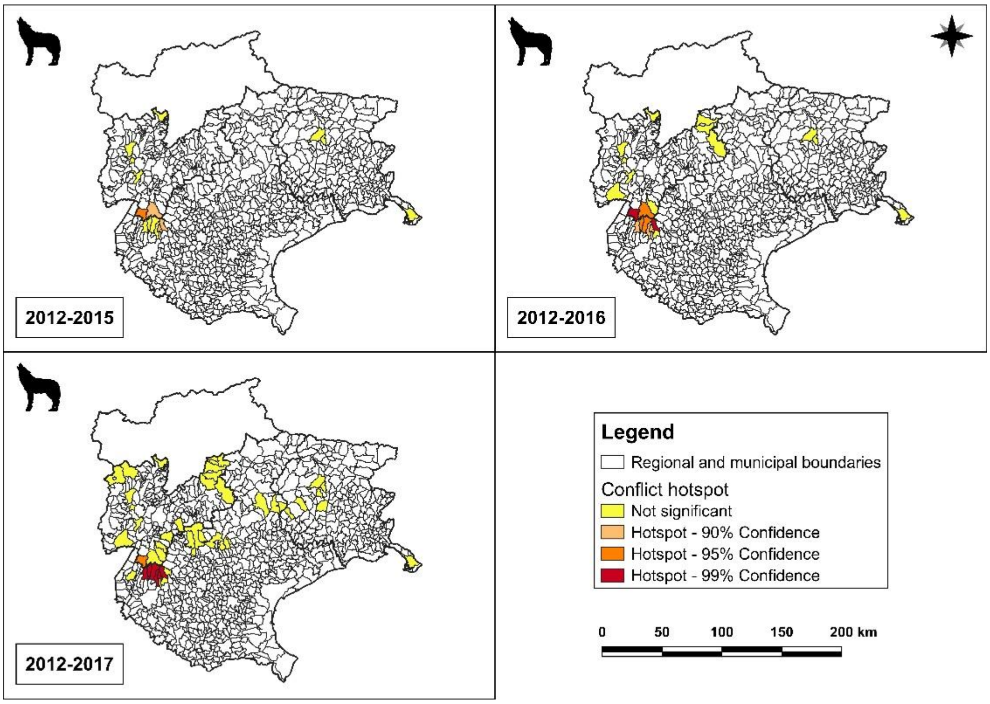
(c)

Figure 2. Maps of human–bear conflict in the study areas from **(a)** 2012 to 2014, **(b)** 2012 to 2017, and **(c)** 2012 to 2020. White polygons indicate those municipalities in which no attacks occurred; yellow polygons indicate non–significant hotspots; light orange polygons indicate hotspots at 90% Confidence Interval (CI); orange polygons indicate hotspots at 95% CI; red polygons indicate hotspots at 99% CI. Brown bear silhouette (https://www.hiclipart.com/free-transparent-background-png-clipart-peawm).

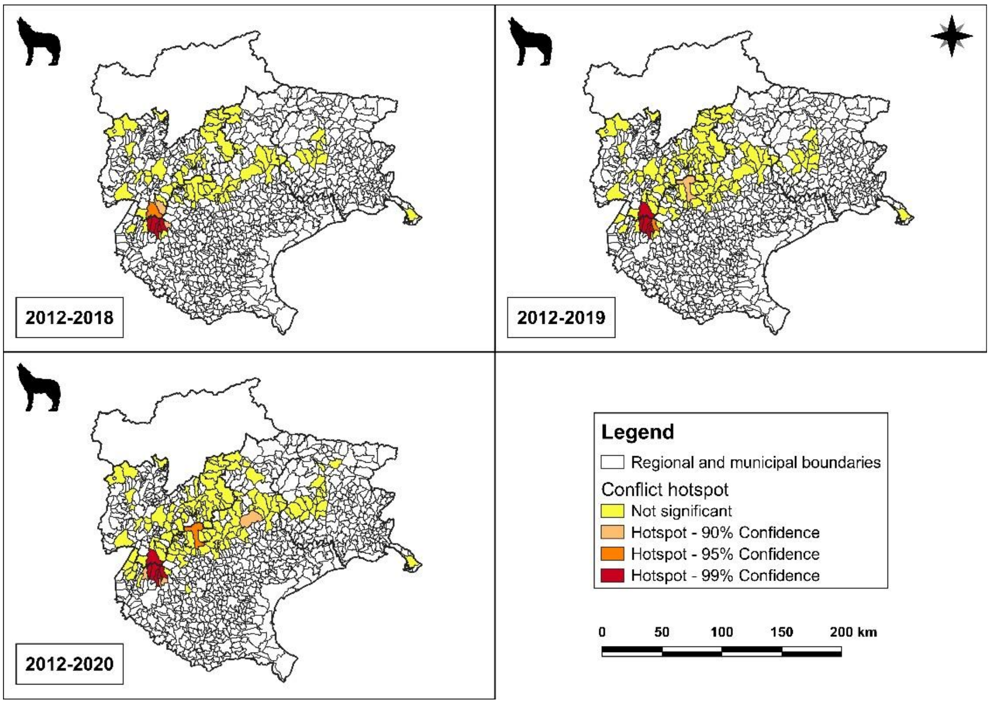
As for wolves, the spatial autocorrelation analyses revealed a clustered data distribution in the latest time intervals (**Appendices 4e–4i**). In fact, no conflict hotspots were observed in the time interval 2012–2014 (**Fig. 3a**). However, in 2012–2016 a hotspot of wolf attacks was observed in the 'Lessinia' highland, a pre–Alpine territory falling in the provinces of Verona and Vicenza (Veneto region), and the southern area of the Autonomous province of Trento (**Fig. 3b**). The same hotspot was observed also in the following time intervals: 2012–2017 (**Fig. 3b**), 2012–2018, 2012–2019, and 2012–2020 (**Fig. 3c**).


2012
2012-2013
2012-2014
Legend
Regional and municipal boundaries
Conflict hotspot
Not significant
Hotspot - 90% Confidence
Hotspot - 95% Confidence
Hotspot - 99% Confidence
0 50 100 150 200 km

(a)


2012-2015
2012-2016
2012-2017
Legend
Regional and municipal boundaries
Conflict hotspot
Not significant
Hotspot - 90% Confidence
Hotspot - 95% Confidence
Hotspot - 99% Confidence
0 50 100 150 200 km

(b)



(c)

Figure 3. Maps of human–wolf conflict in the study areas from **(a)** 2012 to 2014, **(b)** 2012 to 2017, and **(c)** 2012 to 2020. White polygons indicate those municipalities in which no attacks occurred; yellow polygons indicate non–significant hotspots; light orange polygons indicate hotspots at 90% Confidence Interval (CI); orange polygons indicate hotspots at 95% CI; red polygons indicate hotspots at 99% CI. Grey wolf silhouette (https://icon2.kisspng.com/20171221/caq/howling-wolf-silhouette-png-transparent-clip-art-image-5a3bab9caaa7d3.4042705115138599966699.jpg).

As for mountain farms, the spatial autocorrelation analyses revealed a clustered data distribution (**Appendix 5**). Two hotspots of mountain farms in both the 'Asiago' (Veneto region) and 'Lessinia' highland were observed (**Fig. 4**), thus spatially overlapping with the hotspot of wolf attacks observed in the same area (**Fig. 3b, c**).

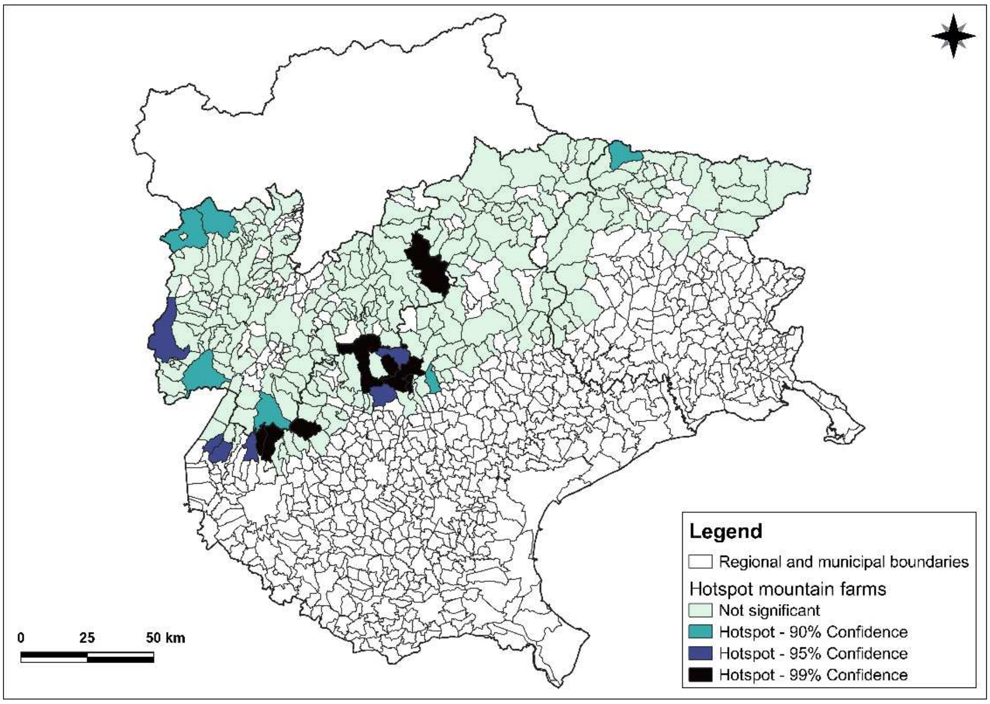


Figure 4. Hotspots of mountain farms in the study areas. White polygons indicate those municipalities in which no mountain farms occur; light blue polygons indicate non–significant hotspots; green water polygons indicate hotspots at 90% Confidence Interval (CI); purple polygons indicate hotspots at 95% CI; black polygons indicate hotspots at 99% CI.

For what concerns the hotspot of livestock abundance, this analysis was realized only considering those municipalities in which wolf attacks occurred (time interval 2012–2020) mainly because of two reasons: (*i*) no hotspots conflictive areas were observed for bears, and (*ii*) we wanted to avoid biasing the results though the inclusions of lowland municipalities in which solely intensive livestock activities are conducted. The spatial autocorrelation analyses revealed a clustered data distribution (**Appendix 6**). Specifically, two hotspots of livestock abundance were observed in the province of Pordenone (FVG region) and in the 'Lessinia' highland (**Fig. 5**), thus spatially overlapping with the hotspot of wolf attacks observed in the same area (**Fig. 3b, c**).

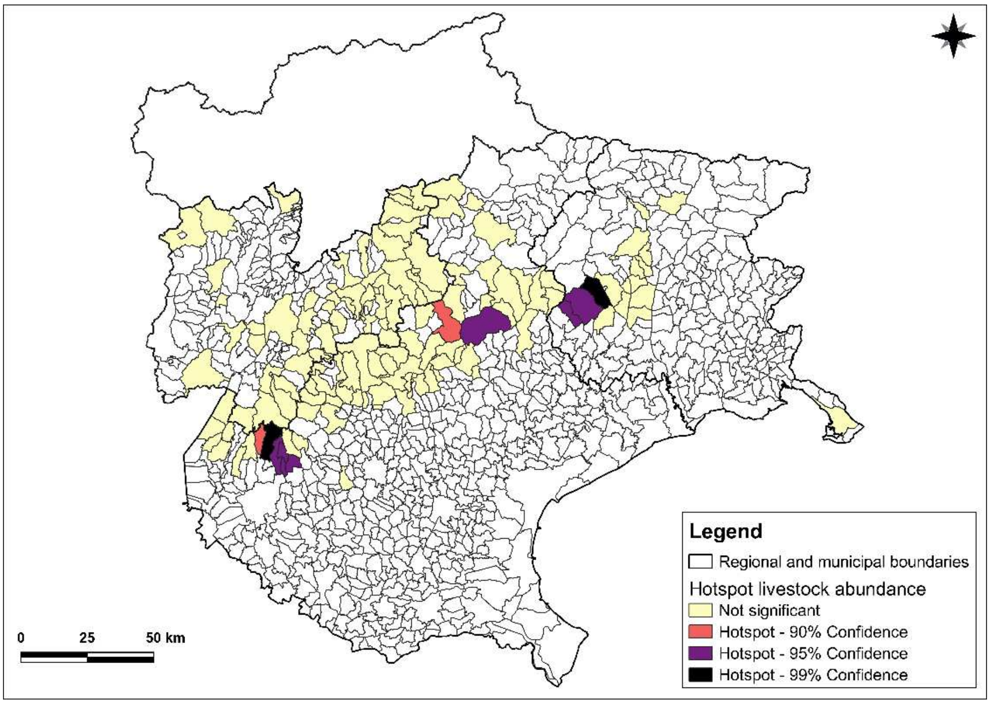


Figure 5. Hotspots of livestock abundance (cattle, sheep and goats) in the study areas. White polygons indicate those municipalities excluded from the analysis; yellow ochre polygons indicate non–significant hotspots; orange polygons indicate hotspots at 90% Confidence Interval (CI); purple polygons indicate hotspots at 95% CI; black polygons indicate hotspots at 99% CI.

5.3.2 Characterization of large carnivores' predations

From 2012 to 2020, n = 327 bear predations were reported: n = 276 (84.40%) in the Autonomous province of Trento, and n = 51 (15.60%) in the FVG region (**Table 1**). Conversely, n = 987 wolf predations were reported: n = 727 (73.66%) in the Veneto region, n = 224 (22.69%) in the Autonomous province of Trento, and n = 36 (3.65%) in the FVG region (**Table 2**). As for bears, we registered n = 271 (82.87%) attacks towards sheep and goats, and n = 56 (17.13%) towards cattle. Conversely, for wolves, we registered n = 547 (55.42%) attacks towards cattle, and n = 440 (44.58%) towards sheep and goats.

Table 1. Number of bear predatory events recorded per region/province, municipality, and year.

Municipality per region/Autonomous province	Number of attacks per year									Total
	2012	2013	2014	2015	2016	2017	2018	2019	2020	
Friuli Venezia Giulia	**10**	**9**	**6**	**2**	**1**	**4**	**5**	**7**	**7**	**51**
Ampezzo	2									2
Dogna			1							1
Enemonzo						1		1		2
Forni di Sopra				1						1
Forni di Sotto					1					1
Lusevera		1	2							3
Malborghetto Valbruna	1									1
Maniago	1									1
Moggio Udinese			2							2
Ovaro		1								1
Pontebba	1	1	1							3
Prato Carnico									1	1
Ravascletto	1	1								2
Resia	4	2				3				9
Sauris		2					1	2	3	8
Socchieve							2	4	1	7
Tarvisio		1							2	3
Torreano							1			1
Tramonti di Sopra				1						1
Verzegnis							1			1
Trento	**39**	**45**	**34**	**12**	**22**	**29**	**56**	**38**	**1**	**276**
Ala		1								1
Altopiano della Vigolana								2		2
Arco	1									1
Avio		3					1			4
Bleggio Superiore							1			1
Bocenago		1	1					1		3
Bondone	1	1								2
Borgo Chiese					1		1			2
Brentonico		9		1	2	1				13
Bresimo		4	1							5
Caderzone Terme	2	2	2		1		3			10
Caldes		2	4	3	2		3			14

Carisolo	1								1
Castello Tesino								1	1
Castello-Molina di Fiemme								3	3
Cavedine		1	4		3	1			9
Cavizzana			1						1
Cles			1				1		2
Comano Terme		1				3	1		5
Denno	2								2
Drena		1	2		1				4
Fiave						1	1		2
Giustino		1					1		2
Grigno			2						2
Ledro					1		2		3
Levico Terme			2						2
Madruzzo		1	1	2					4
Male							1	1	2
Massimeno		1							1
Mezzano	1								1
Mori			1						1
Novella				1					1
Peio				1	1	1			3
Pelugo	1				1	1		2	5
Pieve di Bono-Prezzo						3	1		4
Pinzolo		6	2				1	3	12
Pomarolo	1								1
Porte di Rendena						2	6	4	12
Rabbi	14	2		3		1	3		23
San Lorenzo Dorsino	8	3	3		2	1	1	1	19
Sella Giudicarie			1			3	16	8	28
Spiazzo	1	2			1	3	1	2	10
Stenico		1			1	2	1		5
Strembo	1		1		1			3	6
Telve								1	1
Tenno							1		1
Terzolas			1	1	1	1	2		6
Tione di Trento						3	1	1	5
Tre Ville						1	2		3
Trento	1	2	1		1				5

Valdaone							4	4		8
Vallelaghi	1		2		1	1				5
Villa Lagarina	3									3
Ville d'Anaunia			1		1			1		3
Ville di Fiemme									1	1
Total	**49**	**54**	**40**	**14**	**23**	**33**	**61**	**45**	**8**	**327**

Table 2. Number of wolf predatory events recorded per region/province, municipality, and year.

Municipality per region/Autonomous province	Number of attacks per year									Total
	2012	2013	2014	2015	2016	2017	2018	2019	2020	
Friuli Venezia Giulia		**3**	**4**	**1**		**5**	**6**	**12**	**5**	**36**
Arba							1			1
Aviano									2	2
Budoia						1				1
Castelnovo del Friuli								1		1
Clauzetto			1							1
Enemonzo									1	1
Maniago							3			3
Montereale Valcellina						1				1
San Giorgio della Richinvelda								2		2
San Quirino								2		2
Sequals							1			1
Spilimbergo						1		4	1	6
Tolmezzo									1	1
Tramonti di Sotto		3	2	1						6
Travesio						2				2
Trieste			1							1
Vivaro							1	3		4
Trento		**3**	**7**	**14**	**31**	**48**	**69**	**41**	**11**	**224**
Ala		3	4	10	20	12	16	8	2	75
Altopiano della Vigolana								5		5
Avio				1		1		1		3
Baselga di Pine								1		1
Borgo d'Anaunia				1						1
Campitello di Fassa						5	5			10
Canal San Bovo								2		2

Canazei				6	1	2		9
Castel Ivano						1		1
Castello Tesino						2		2
Castello-Molina di Fiemme						1		1
Cavedine					2			2
Cinte Tesino						1		1
Comano Terme		2	1			1		4
Fierozzo						1		1
Folgaria				1	2	1		4
Grigno					1			1
Imer							2	2
Lavarone					1			1
Ledro			1					1
Levico Terme				1	6	3		10
Luserna				1				1
Madruzzo					2			2
Mazzin				1				1
Mezzano					1		2	3
Moena			1	3	4			8
Novaledo					2			2
Palu del Fersina						1		1
Peio				8	1	1	1	11
Pergine Valsugana						1		1
Pieve Tesino					2			2
Pinzolo	3							3
Predazzo				5	2	1		8
Primiero San Martino di Castrozza			6		2	2	2	12
Rabbi				2	1	1	1	5
Roncegno Terme					2			2
Rovereto						1		1
San Giovanni di Fassa			1		5			6
Scurelle					1			1
Telve					3	1		4
Telve di Sopra					2			2
Tesero							1	1
Trambileno				1	1	1		3
Trento					1			1
Vallarsa			1	1	3	1		6

Veneto	7	10	30	39	34	110	152	129	216	727
Alano di Piave							1	1		2
Alpago						1			13	14
Arsie								1		1
Arsiero							3	1	1	5
Asiago						7	8	8	13	36
Badia Calavena					1			4	3	8
Bassano del Grappa								2		2
Belluno						1	4	4	10	19
Borgo Valbelluna							14	9	18	41
Borso del Grappa						3	1			4
Bosco Chiesanuova	3	10	14	19	14	37	24	16	30	167
Brenzone sul Garda								2	1	3
Caltrano						1				1
Calvene								2	1	3
Caprino Veronese						1			1	2
Cesiomaggiore									1	1
Chies d'Alpago									11	11
Cogollo del Cengio									1	1
Crespadoro						2	1	1		4
Enego							12	8	6	26
Erbezzo	2		10	7	3	3	4	2	7	38
Feltre									5	5
Ferrara di Monte Baldo									5	5
Foza							2		3	5
Fumane									1	1
Gallio						2	12	10	4	28
Gosaldo							1		1	2
Laghi									1	1
Lastebasse									1	1
Limana							1		1	2
Livinallongo del Col di Lana							1	1		2
Lusiana Conco								7	6	13
Malcesine									1	1
Montecchio Maggiore									1	1
Pieve del Grappa						1	1	1	1	4
Ponte nelle Alpi							2		3	5
Posina									1	1

Possagno						1	1			2
Pove del Grappa							5			5
Quero Vas							2	1	1	4
Roana						2	17	5	3	27
Rocca Pietore							4	1	1	6
Rotzo							2	1		3
Rovere Veronese			2	3	4	11	2	1	12	35
Salcedo								1		1
San Mauro di Saline						1			1	2
San Zeno di Montagna									1	1
Sant'Anna d'Alfaedo			1	3	1	4	3	2	11	25
Schio								3	1	4
Sedico								1	1	2
Segusino							1	2	1	4
Selva di Cadore									1	1
Selva di Progno				1	7	21	9	7		45
Seren del Grappa							2	1	1	4
Solagna							2	1	1	4
Sovramonte								1	2	3
Tambre						1			4	5
Val di Zoldo									2	2
Valbrenta						3	6	5	5	19
Valdobbiadene								1		1
Valli del Pasubio								1		1
Velo d'Astico									3	3
Velo Veronese	2		3	6	4	7	3	13	11	49
Vidor								1		1
Vittorio Veneto							1		1	2
Total	**7**	**16**	**41**	**54**	**65**	**163**	**227**	**182**	**232**	**987**

As for bear predations, from the set of GLMMs it turned out that the best model was the one in which the effect of the covariate *livestock* was considered (**Table 3**).

Table 3. Generalized linear mixed models (GLMMs) ranking referring to bear predations. The best model was *italicized*. Abbreviations: NDVI = normalized difference vegetation index; liv_density = livestock density (n. ind./ha); farms = number of mountain farms; abundance_min = minimum bear abundance;

livestock = affected livestock species (i.e., cattle, sheep and goats); AIC = Akaike's information criterion; ωi = Akaike's weight. Interactions were denoted by ":".

Model ID	Fixed factors	Random factor	AIC	ΔAIC	ωi
1	*livestock*	Municipality	*1187.63*	*0.00*	*0.60*
2	year + livestock		1188.45	0.82	0.40
3	year + liv_dens + livestock		1198.74	11.10	0.00
4	year + NDVI + liv_dens + livestock		1200.73	13.10	0.00
5	year:NDVI + year + NDVI + liv_dens + livestock		1208.20	20.57	0.00
6	year:NDVI + period + year + NDVI + liv_dens + livestock		1209.72	22.09	0.00
7	year:NDVI + period + year + NDVI + liv_dens + abundance_min + livestock		1211.19	23.55	0.00
8	year:NDVI + period + year + NDVI + liv_dens + farms + abundance_min + livestock		1213.15	25.52	0.00

This model showed no significant effect (β = 0.39, SE = 0.20, p = 0.05) of the covariate *livestock*.

As for wolf predations, from the set of GLMMs it turned out that the best model was the one in which the effect of the factor *period* and covariate *livestock* was considered (**Table 4**).

Table 4. Generalized linear mixed models (GLMMs) ranking referring to wolf predations. The best model was *italicized*. Abbreviations: liv_density = livestock density (n. ind./ha); pasture = surface covered by each pasture (ha); farms = number of mountain farms; abundance_min = minimum wolf abundance; livestock = affected livestock species (i.e., cattle, sheep and goats); AIC = Akaike's information criterion; ωi = Akaike's weight.

Model ID	Fixed factors	Random factor	AIC	ΔAIC	ωi
1	*period + livestock*	Municipality	*3656.02*	*0.00*	*0.30*
2	period + abundance_min + livestock		3656.22	0.21	0.27
3	period + liv_dens + abundance_min + livestock		3656.39	0.37	0.25
4	period + liv_dens + pastures + abundance_min + livestock		3657.84	1.82	0.12
5	period + liv_dens + pastures + farms + abundance_min + livestock		3659.38	3.36	0.06

This model showed that the probability to observe an increasing number of livestock predated/wounded by wolves was significantly higher (β = 0.23, SE = 0.11, p = 0.03) of about 0.23 units during the transhumant period compared to the non–transhumant one. Furthermore, compared to cattle, the probability to observe an increasing number of sheep and goats predated/wounded by wolves was significantly higher ($\beta = 1.32$, SE = 0.08, $p < 0.001$) of about 1.32 units.

Most wolf attacks in the Veneto region from 2018 to 2020 occurred during the night (n = 253, 62.47%) followed by early in the morning (n = 121, 29.88%), day (n = 20, 4.94%), evening (n = 8, 1.98%), late in the morning (n = 2, 0.49%), and late afternoon (n = 1, 0.25%) (**Table 5a**). A significant difference (Fisher's test, $p < 0.001$) in terms of number of attacks was recorded between daily periods. Specifically, a significant difference was found comparing 'early in the morning' vs 'late in the morning' (*pnif*, $p < 0.001$), 'early in the morning' vs 'day' (*pnif*, $p < 0.001$), 'early in the morning' vs 'late afternoon' (*pnif*, $p < 0.001$), 'early in the morning' vs 'evening' (*pnif*, $p < 0.001$), 'early in the morning' vs 'night' (*pnif*, $p < 0.001$), 'late in the morning' vs 'day' (*pnif*, $p < 0.001$), 'late in the morning' vs 'night' (*pnif*, $p < 0.001$), 'day' vs 'late afternoon' (*pnif*, $p < 0.001$), 'day' vs 'evening' (*pnif*, $p = 0.04$), 'day' vs 'night' (*pnif*, $p < 0.001$), 'late afternoon' vs 'evening' (*pnif*, $p = 0.04$), 'late afternoon' vs 'night' (*pnif*, $p < 0.001$), and 'evening' vs 'night' (*pnif*, $p < 0.001$). Conversely, no significant difference was found comparing 'late in the morning' vs 'late afternoon' (*pnif*, $p = 1.00$), and 'late in the morning' vs 'evening' (*pnif*, $p = 0.11$).

The highest number of wolf attacks in the Veneto region from 2018 to 2020 was recorded in open pastures (n = 335, 74.78%), followed by canopy–covered areas (n = 97, 21.65%), other (n = 14, 3.13%), fragmented pastures (n = 1, 0.22%), and stable (n = 1, 0.22%) (**Table 5b**). A significant difference (Fisher's test, $p < 0.001$) in terms of number of attacks was recorded between daily periods. Specifically, a significant difference was found comparing 'canopy–covered areas' vs 'fragmented pastures' (*pnif*, $p < 0.001$), 'canopy–covered areas' vs 'open pastures' (*pnif*, $p < 0.001$), 'canopy–covered areas' vs 'stable' (*pnif*, $p < 0.001$), 'canopy–covered areas' vs 'other' (*pnif*, $p < 0.001$), 'fragmented pastures' vs 'open pastures' (*pnif*, $p < 0.001$), 'fragmented pastures' vs 'other' (*pnif*, $p < 0.001$), 'open pastures' vs 'stable' (*pnif*, $p < 0.001$), 'open pastures' vs 'other' (*pnif*, $p < 0.001$), and 'stable' vs 'other' (*pnif*, $p < 0.001$). On the contrary, no significant difference was found comparing 'fragmented pastures' vs 'stable' (*pnif*, $p = 1.00$).

Most wolf attacks in the Veneto region from 2018 to 2020 occurred at a distance 0–150 m from the nearest human building (n = 229, 53.88%), followed by 151–300 m (n = 84, 19.76%), 451–600 m (n = 53, 12.47%), > 600 m (n = 31, 7.29%), and 301–450 m (n = 28, 6.59%) (**Table 5c**). A significant difference ($\chi^2 = 410.68$, $p < 0.001$) in terms of number of attacks was recorded between distances from the nearest building. Specifically, a significant difference was found comparing '0–150 m' vs '151–300 m' (*pnif*, $p < 0.001$), '0–150 m' vs '301–450 m' (*pnif*, $p < 0.001$), '0–150 m' vs '451–600 m' (*pnif*, $p < 0.001$), '0–150 m' vs '> 600 m' (*pnif*, $p < 0.001$), '151–300 m' vs '301–450 m' (*pnif*, $p < 0.001$), '151–300 m' vs '451–600 m' (*pnif*, $p = 0.006$), '151–300 m' vs '> 600 m' (*pnif*, $p < 0.001$), '301–450 m' vs '451–600 m' (*pnif*, $p = 0.006$), and '451–600 m' vs '> 600 m' (*pnif*, $p = 0.02$). On the contrary, no significant difference was found comparing '301–450 m' vs '> 600 m' (*pnif*, $p = 0.79$).

The majority of wolf attacks in the Veneto region from 2018 to 2020 occurred during days of clear weather (n = 296, 70.31%), followed by raining/stormy days (n = 93, 22.09%), foggy days (n = 23, 5.46%), cloudy days (n = 5, 1.19%), and snowy days (n = 4, 0.95%) (**Table 5d**). A significant difference (Fisher's test, $p < 0.001$) in terms of number of attacks was recorded in relation to daily climatic conditions. Specifically, a significant difference was found comparing 'days of clear weather' vs 'raining/stormy days' (*pnif*, $p < 0.001$), 'days of clear weather' vs 'cloudy days' (*pnif*, $p < 0.001$), 'days of clear weather' vs 'foggy days' (*pnif*, $p < 0.001$), 'days of clear weather' vs 'snowy days' (*pnif*, $p < 0.001$), 'raining/stormy days' vs 'cloudy days' (*pnif*, $p < 0.001$), 'raining/stormy days' vs 'foggy days' (*pnif*, $p < 0.001$), 'raining/stormy days' vs 'snowy days' (*pnif*, $p < 0.001$), 'cloudy days' vs 'foggy days' (*pnif*, $p < 0.001$), and 'foggy days' vs 'snowy days' (*pnif*, $p < 0.001$). Conversely, no significant difference was found comparing 'cloudy days' vs 'snowy days' (*pnif*, $p = 1.00$).

The majority of wolf attacks in the Veneto region from 2018 to 2020 occurred towards cattle of 6–12 months of age (n = 71, 40.80%), followed by < 6 months (n = 46, 26.44%), 12–18 months (n = 38, 21.84%), 18–24 months (n = 10, 5.75%), and > 24 months (n = 9, 5.17%) (**Table 5e**). A significant difference ($\chi^2 = 97.94$, $p < 0.001$) in terms of number of attacks was recorded between the different attacked age classes. Specifically, a significant difference was found comparing '< 6 months' vs '6–12 months' (*pnif*, $p = 0.008$),

‘< 6 months’ vs ‘18–24 months’ (*pnif*, $p < 0.001$), ‘< 6 months’ vs ‘> 24 months’ (*pnif*, $p < 0.001$), ‘6–12 months’ vs 12–18 months’ (*pnif*, $p < 0.001$), ‘6–12 months’ vs ‘18–24 months’ (*pnif*, $p < 0.001$), ‘6–12 months’ vs ‘> 24 months’ (*pnif*, $p < 0.001$), ‘12–18 months’ vs ‘18–24 months’ (*pnif*, $p < 0.001$), and ‘12–18 months’ vs ‘> 24 months’ (*pnif*, $p < 0.001$). Conversely, no significant difference was found comparing ‘< 6 months’ vs ‘> 12–18 months’ (*pnif*, $p = 0.42$) and ‘18–24 months’ vs ‘> 24 months’ (*pnif*, $p = 1.00$).

Table 5. Overall number of wolf attacks in the Veneto region from 2018 to 2020 in relation to **(a)** period of the day, **(b)** habitat, **(c)** distance from the nearest human building, **(d)** climatic conditions, and **(e)** age class of cattle.

		Number of attacks
(a) Period of the day	Early in the morning	121
	Late in the morning	2
	Day	20
	Late afternoon	1
	Evening	8
	Night	253
(b) Habitat	Canopy-covered	97
	Fragmented pasture	1
	Open pasture	335
	Stable	1
	Other	14
(c) Distance from the nearest building	0-150	229
	151-300	84
	301-450	28
	451-600	53
	> 600	31
(d) Climatic conditions	Clear weather	296
	Raining/Storm	93
	Cloudy	5
	Foggy	23
	Snowy	4
(e) Age class (cattle)	< 6	46
	6-12	71

	12-18	38
	18-24	10
	> 24	9

5.4 Discussion

The results obtained from the hotspot analyses confirmed our initial assumption. In fact, a single hotspot of human–wolf negative interaction was observed in the 'Lessinia' highland, while no hotspots of human–bear conflict were observed. As for wolves, a spatial overlap was furtherly observed between hotspot of wolf attacks and hotspot of both mountain farms and/or livestock abundance in the 'Lessinia' highland. These findings suggest that the different feeding behaviour of these predators (e.g., Bojarska & Selva 2012; Zlatanova *et al*. 2014), may bias the degree of negative interactions towards wolves rather than bears. In spite of these considerations, the presence of single or hotspots of carnivore attacks suggests that livestock management practices and/or the habitat conditions in these areas may increase the risk of negative interactions. For instance, in the 'Lessinia' highland grazing is organized with the purpose to minimize labour and costs. Therefore, especially in the last decade, livestock was usually left unattended during the day and night in unprotected pastures, with consequent higher risks of wolf predations (Faccioni *et al*. 2015).

As for bears, the results obtained from the best model revealed that the probability of predation was not significantly different comparing cattle with sheep and goats. This result is inconsistent with other researches realized in Italy (e.g., Fico *et al*. 1993) and other European countries (e.g., Slovenia – Kavčič *et al*. 2013; Croatia – Hipólito *et al*. 2020) in which it was shown that sheep are amongst the preferred domestic prey in the area, mainly because of their poor anti–predatory behaviour. Therefore, this finding may have been biased by the availability of each livestock species (which may vary depending on the area), which in turn affected the probability of attacks towards certain categories. Contrary to our expectations, the probability of bear attacks towards livestock did not reduce in those years characterized by higher vegetation productivity and/or in response to higher bear abundance. This finding may thus find explanation in the behaviour of each individual which could lead bears to be more prone to predate on livestock, especially if the respective mother was a “problematic” bear (Franchini *et al*. 2020; Bombieri *et al*. 2021). As for wolves, the findings obtained from the best model showed that the probability of predation was significantly higher during the transhumant period. These results are consistent with those obtained by Faccioni *et al*. (2015) in which they observed that wolf predations peaked during summer, where livestock farming predominates in Alpine areas using summer pastures. Moreover, the model also has shown that the probability of predation was significantly higher for sheep and goats than for cattle. Despite sheep and goats could be preferred by wolves because of their smaller size and poor anti–predatory strategies (e.g., Fico *et al*. 1993), these results are partially in accordance with those obtained by Dondina *et al*. (2015). In fact, during their research they observed that the number of killed animals per event was higher for sheep than for cattle. However, the majority of predations were recorded upon cattle. On the same line, Faccioni *et al*. (2015) showed a higher number of predation towards cattle in the ‘Lessinia’ highland, where cattle is the most frequently bred livestock species. These results hence suggest that the availability of each livestock species within an area may drive the impact of wolves towards certain categories. Contrary to our expectations, the results of the model revealed that the overall number of predated/wounded individuals did not increase in response to higher wolf abundance. Like for bears, this result indicates that, regardless of the density, some individuals may be more prone to predate upon livestock compared to others, most likely depending on environmental factors (Dondina *et al*. 2015) and/or livestock management practices (Faccioni et al. 2015).

The highest number of wolf attacks towards livestock was reported during the night. These results are in line with those obtained by Ciucci & Boitani (1998) and Rehman *et al*. (2021) where they observed that wolf predations at night were higher than during the day, most likely reflecting the nocturnal behaviour of the species which may take advantage of the night cover during hunting. The highest number of wolf attacks was registered in open pastures, thus in line with the presence of livestock in this habitat during the summer months. This finding is consistent with Faccioni’s *et al*. (2015) and Dondina’s *et al*. (2015) studies. Specifically, Dondina *et al*. (2015) observed that the majority of wolf attacks were linked to livestock births occurring in pastures. Furthermore, even the shape complexity of pastures due to the increasing edges between pastures and forests can make surveillance and predator detection by herds more difficult (Dondina *et al*. 2015). Lastly, the percentage of suitable habitats for wolves inside and around the pasture may increase the incidence of predations because of the increasing probability of encounters between wolf and livestock (Marucco & McIntire 2010; Dondina *et al*. 2015). However, these environmental variables were not tested in this study, thus calling for the need of further investigations.

The highest number of predatory events recorded during days of clear weather is in line with the findings obtained by Ciucci & Boitani (1998), but contrary to what observed by Iliopoulos *et al*. (2009) which did not observe any difference in terms of livestock attacks in relation to weather conditions. Bad weather conditions directly influence livestock management, decrease the tendency of aggregation, and increase the difficulty of housing, thus contributing to increase the odds of predations (Tropini 2005). Because of these risks, we could speculate that livestock owners may have chosen to concentrate livestock grazing practices mainly during days of clear weather.

Most wolf attacks were observed at a distance varying from 0 to 150 m from the nearest human building, hence reflecting the tendency of wolves to get close human areas in search for food. This finding may find explanation in the period of the day in which most attacks occurred, i.e., night. In fact, during the night–time period wolves are more active (e.g., Rehman *et al*. 2021) while human activities are reduced. Therefore, because of the reduced human disturbance, wolves may be more incline to roam even in the near proximity of human buildings.

Most wolf attacks occurred towards cattle of 6–12 months of age. This result is in line with Faccioni *et al*. (2015) where they observed that the majority of attacks were oriented towards calves < 1 year old and may be explained by the easier catchability of this age class compared to older individuals.

5.5 Research limits

Although our hotspot analysis performed well in mapping bear and wolf damages risk, we are aware about the potential limitations: (*i*) damages by large carnivores are influenced by a set of environmental factors (e.g., Davie *et al*. 2014; Zarzo–Arias *et al*. 2021) which were not included in this analysis. In fact, different models should have been used (e.g., Khosravi *et al*. 2022). However, our goal was to provide a general picture of the problem by using methods of spatial analysis that do not allow to include covariates and/or factors; (*ii*) because drivers of predation risk may be species–specific (Milanesi *et al*. 2019), developing a separate hotspot analyses for each livestock species would have helped to formulate mitigation solutions targeting each livestock. Nevertheless, in our case, the results obtained would have been inconsistent, since the information referring to the overall abundance of sheep and goats at a municipal level provided by the BDN (https://www.vetinfo.it/) were available only starting from 2017 onwards; (*iii*) large carnivores predations towards livestock are also affected by the abundance of wild prey (e.g., Khorozyan *et al*. 2015) and/or implementation of proper prevention measures (e.g., Berzi *et al*. 2021). Nevertheless, these information were absent, incomplete, or rarely reported in the original database.

5.6 Conservation and management implications

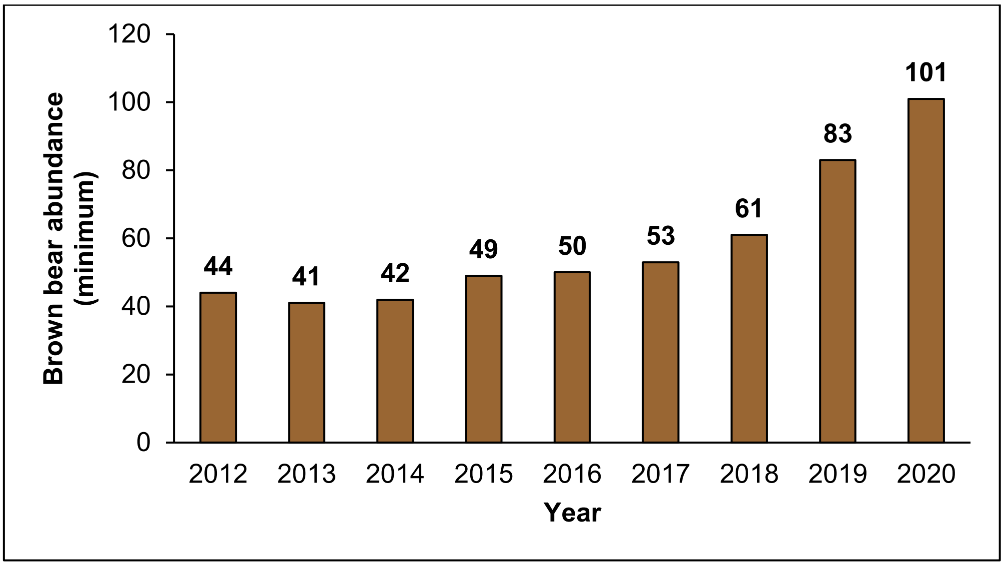
Our research provides important preliminary information for prioritizing management and conservation efforts, particularly in hotspot conflictive areas. Because livestock farming significantly contributes to the rural economy in the Italian Alps, reducing damage risks through efficient and spatially precise conflict mitigations strategies can strengthen both the economic livelihood and biodiversity preservation.

Compensations for damages caused by large carnivores, as well as investments for the realization of prevention measures, have already been used in our study areas to mitigate the impact that carnivores may have on livestock activities. In the Autonomous province of Trento, compensations and funds dedicated to prevention measures are granted based on the Article 33 bis of the Provincial Law n. 24 dated 09/12/1991. In the FVG region, compensations and funds dedicated to prevention measures are delivered based on the Articles 10 and 39 of the Regional Law n. 6 dated 06/03/2008. In the Veneto region, compensations and funds dedicated to prevention measures are granted based on a regional fund (Article 28 of the Regional Law n. 50 dated 1993) and another fund for damages caused by wildlife to both livestock and agricultural activities in those territories in which hunting is forbidden (Article n. 3 c. 1 of the Regional Law n. 6 dated

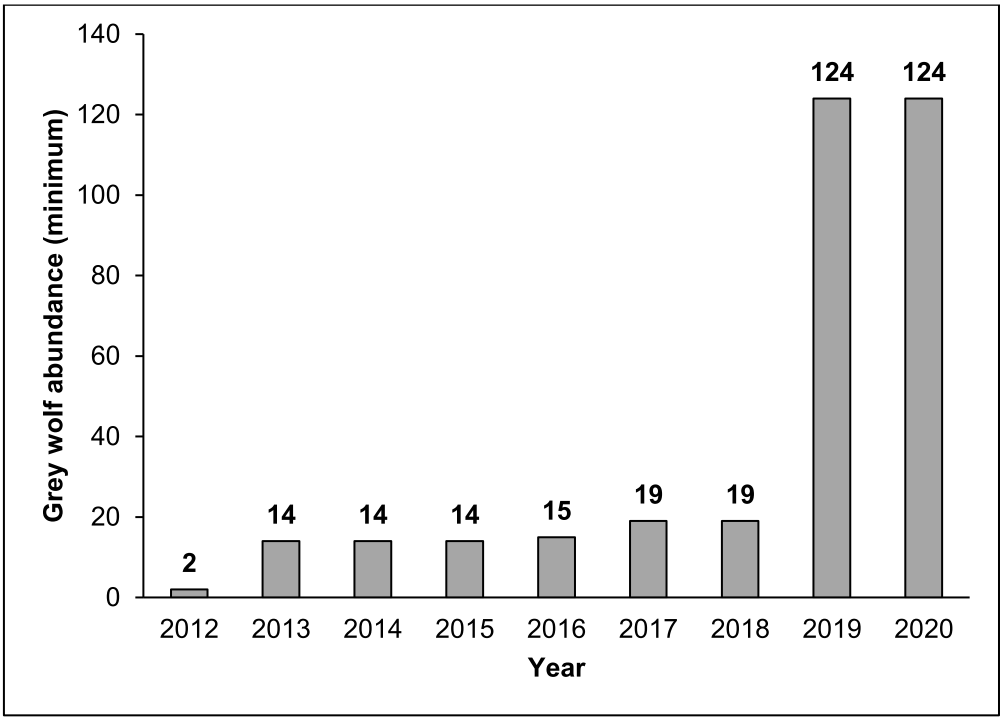
2013) (Franchini *et al*. 2020). However, an unidentified number of declarations of damages is not realized by farmers/livestock owners because of the complicated bureaucracy, improper evaluation of indirect damages, and long–time required to obtain compensations (Franchini *pers. comm.*). Following these considerations, finding solutions aimed at giving proper support to farmers/livestock owners to encourage the coexistence between large carnivores and humans in the Italian Alps assumes thus remarkable importance.

Based on our findings we suggest (*i*) trying to simplify the bureaucratic procedure to obtain compensations, also taking into consideration eventual indirect damages; (*ii*) implementing proper prevention measures (e.g., fences, livestock guarding dogs), especially in hotspot conflictive areas and during the night when carnivore attacks are more prone to occur; (*iii*) providing adequate technical support to farmers/livestock owners, especially during the realization of prevention measures (e.g., fences); (*iv*) avoid concentrating livestock births in open pastures to minimize the risks of carnivore attacks; (*v*) creating a sort of 'landscape of fear' for predators using non–lethal weapons (e.g., rubber bullets) to discourage them to get close human areas; and (*vi*) as reported in the Article 16.1 of the European 'Habitats' Directive 92/43/EEC (Council of the European Union 1992), when the conservation status of the target species is not considered as 'Threatened', derogating to the strict protection regime for bears and wolves in those cases where there is no satisfactory alternative to prevent serious damages, in particular to livestock.

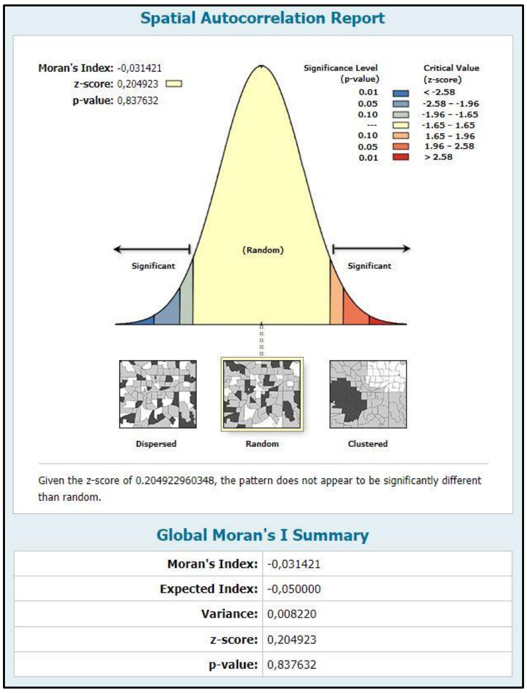
5.7 Appendices



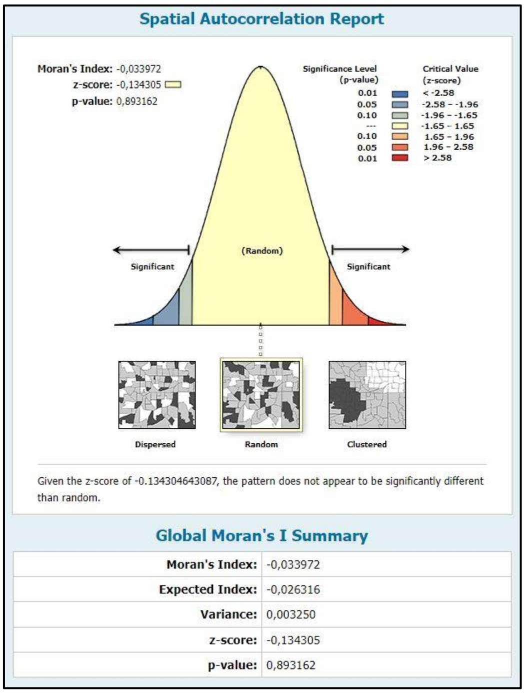
Appendix 1. Minimum number of bears genetically identified within the FVG region and the Autonomous province of Trento per year.



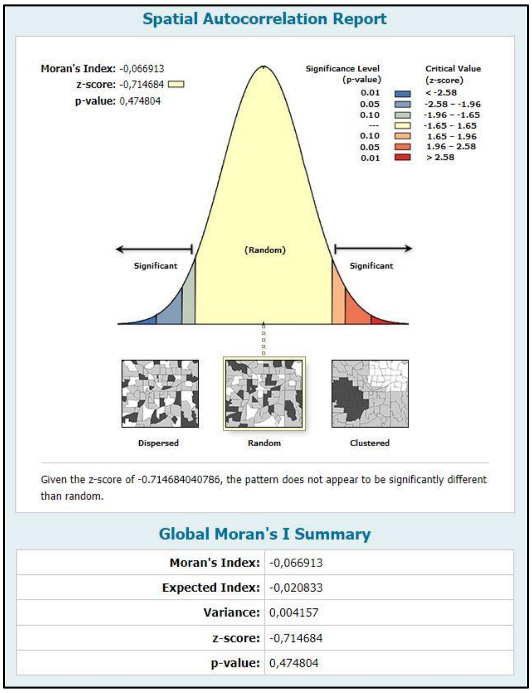
Appendix 2. Minimum number of wolves estimated and/or genetically identified within the whole study area (FVG, Veneto, Autonomous province of Trento) per year.



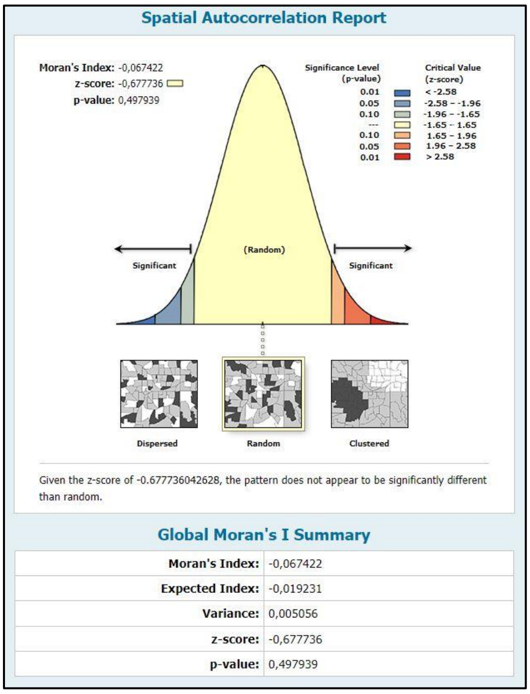
Appendix 3a. Spatial autocorrelation of bear attack data (year 2012).



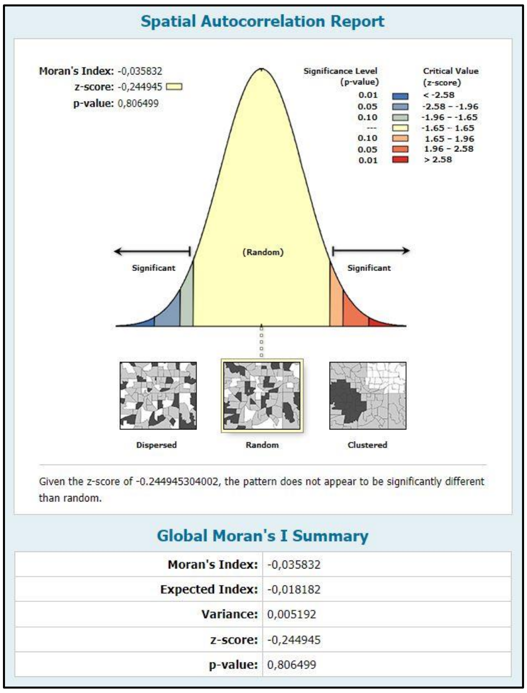
Appendix 3b. Spatial autocorrelation of bear attack data (years 2012–2013).



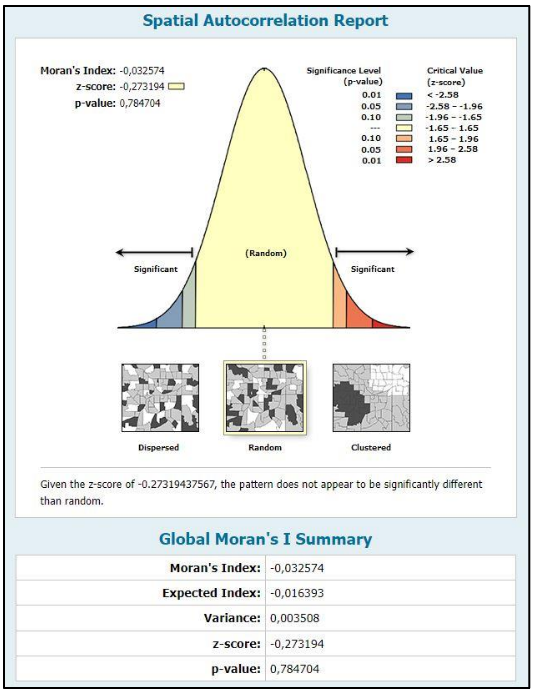
Appendix 3c. Spatial autocorrelation of bear attack data (years 2012–2014).



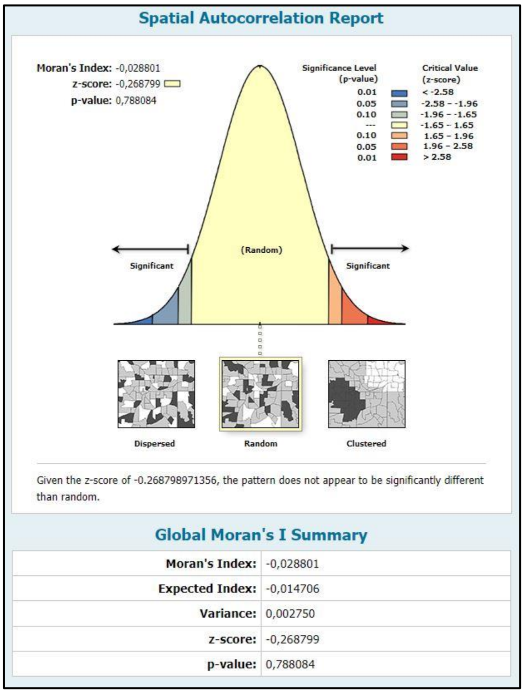
Appendix 3d. Spatial autocorrelation of bear attack data (years 2012–2015).



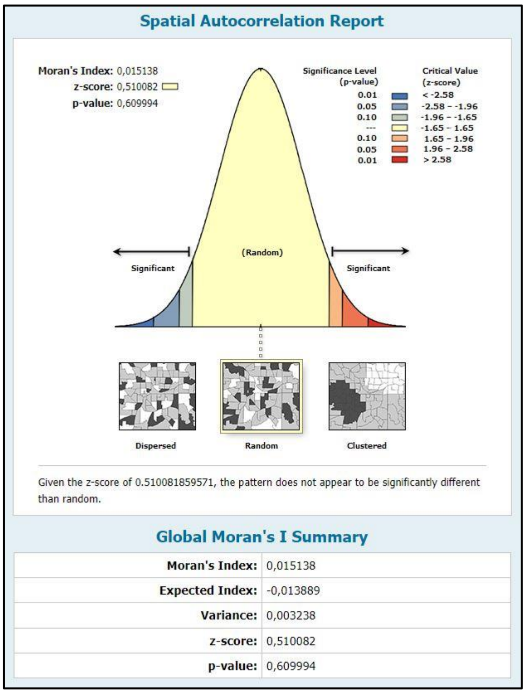
Appendix 3e. Spatial autocorrelation of bear attack data (years 2012–2016).



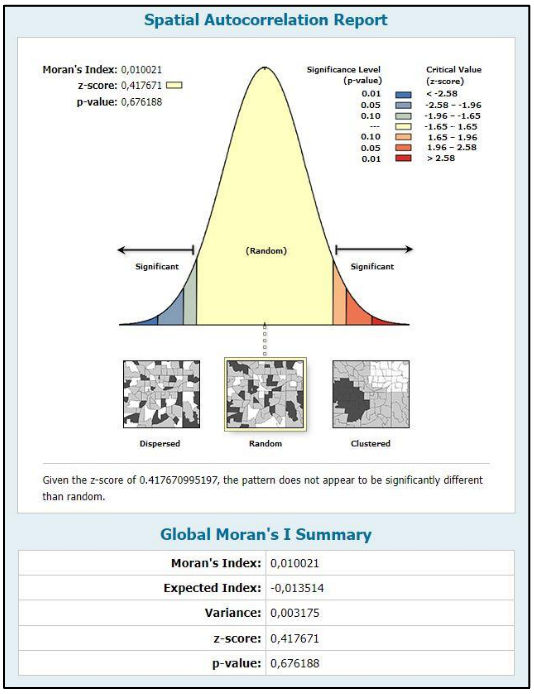
Appendix 3f. Spatial autocorrelation of bear attack data (years 2012–2017).



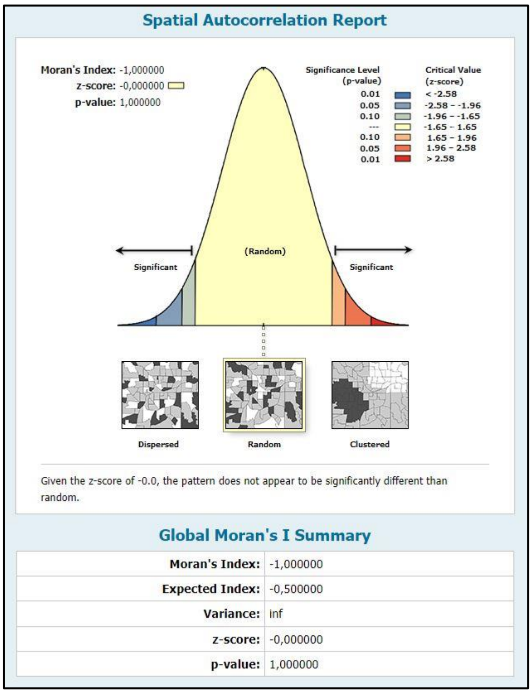
Appendix 3g. Spatial autocorrelation of bear attack data (years 2012–2018).



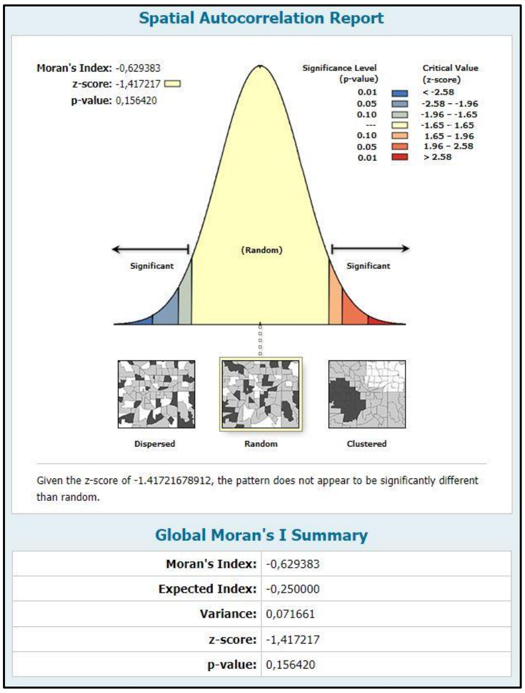
Appendix 3h. Spatial autocorrelation of bear attack data (years 2012–2019).



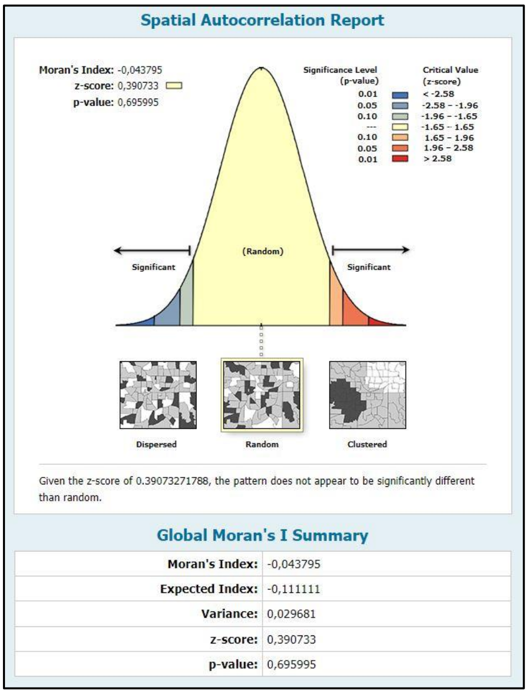
Appendix 3i. Spatial autocorrelation of bear attack data (years 2012–2020).



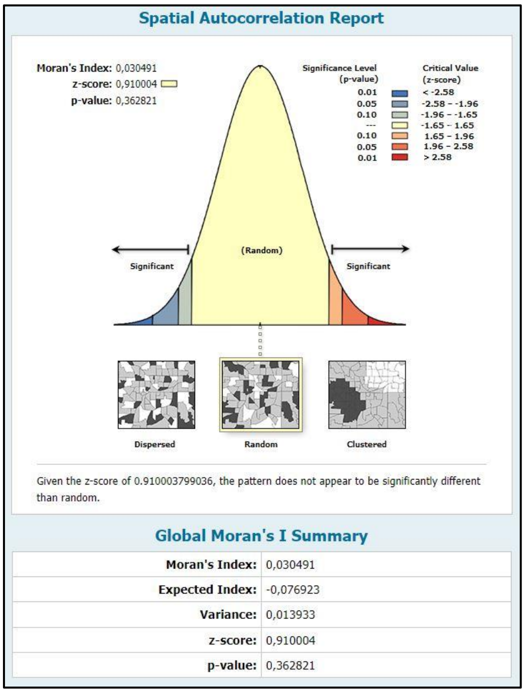
Appendix 4a. Spatial autocorrelation of wolf attack data (year 2012).



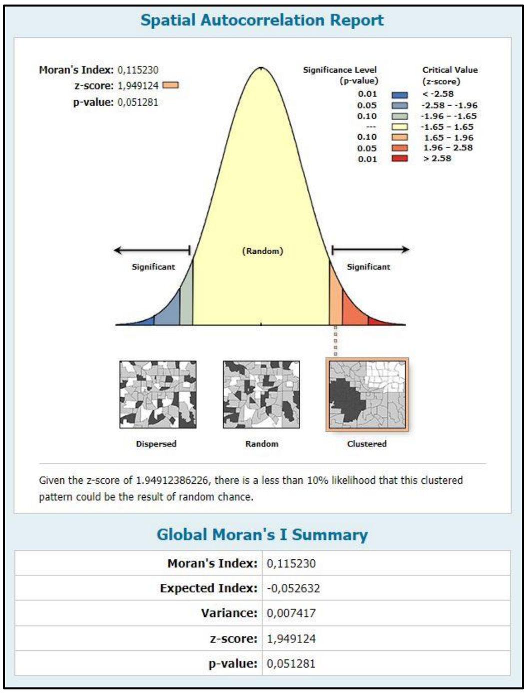
Appendix 4b. Spatial autocorrelation of wolf attack data (year 2012–2013).



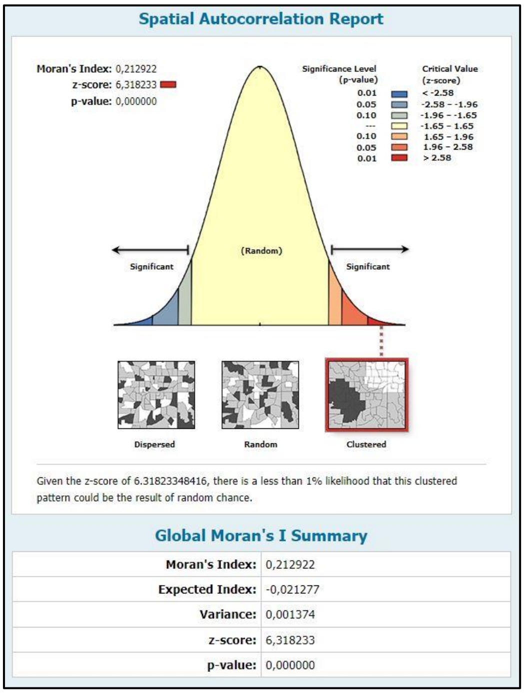
Appendix 4c. Spatial autocorrelation of wolf attack data (year 2012–2014).



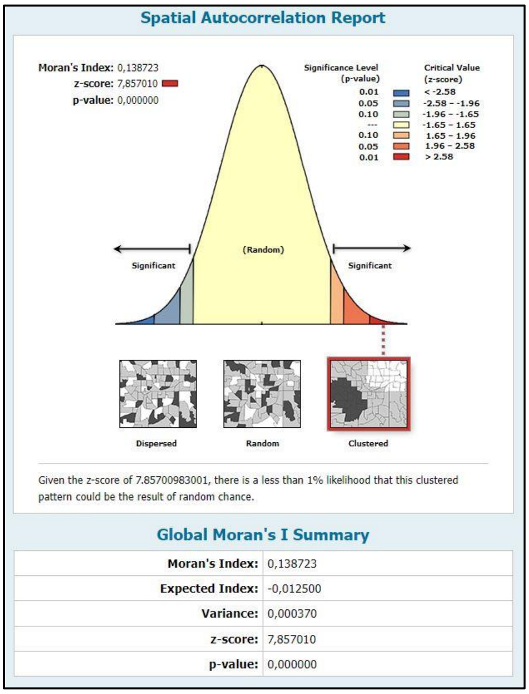
Appendix 4d. Spatial autocorrelation of wolf attack data (year 2012–2015).



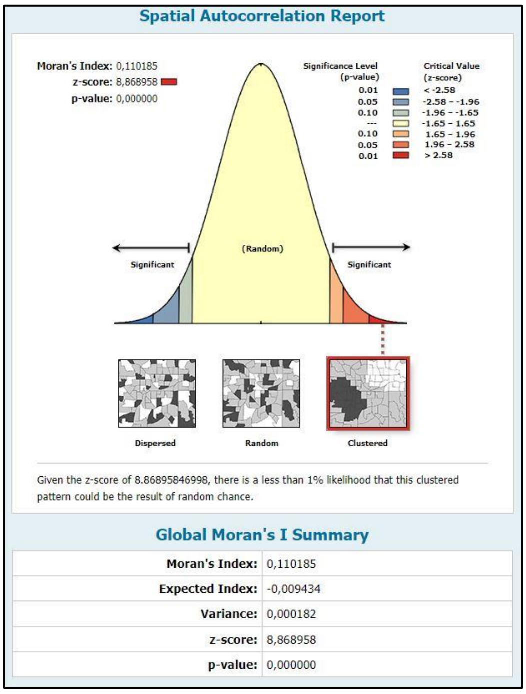
Appendix 4e. Spatial autocorrelation of wolf attack data (year 2012–2016).



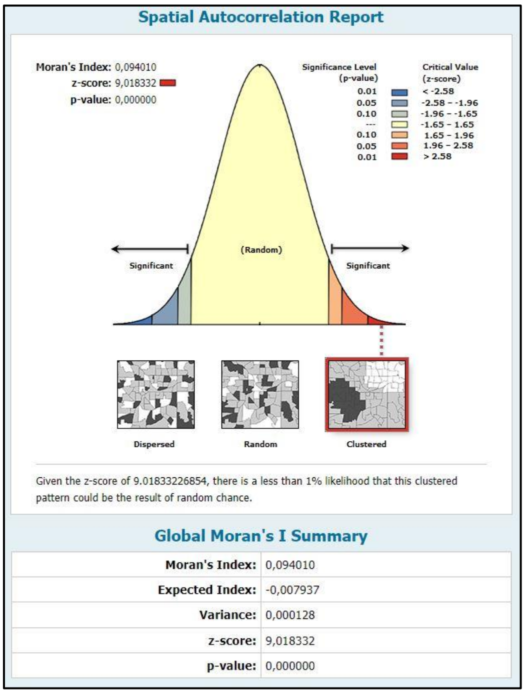
Appendix 4f. Spatial autocorrelation of wolf attack data (year 2012–2017).



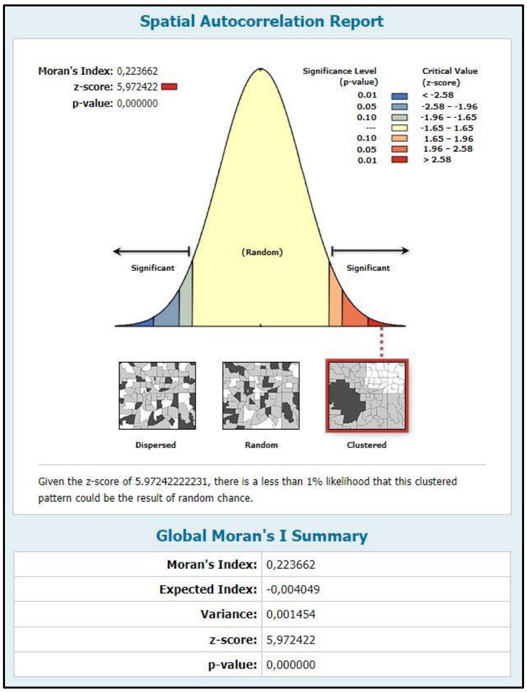
Appendix 4g. Spatial autocorrelation of wolf attack data (year 2012–2018).



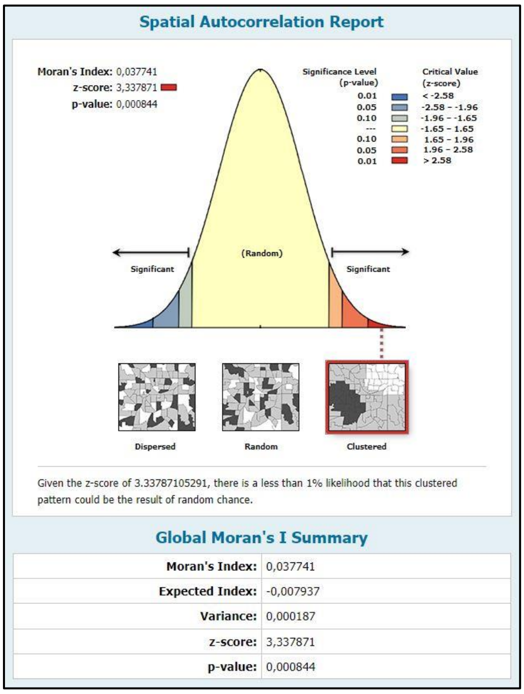
Appendix 4h. Spatial autocorrelation of wolf attack data (year 2012–2019).



Appendix 4i. Spatial autocorrelation of wolf attack data (year 2012–2020).



Appendix 5. Spatial autocorrelation of mountain farm data.



Appendix 6. Spatial autocorrelation of livestock abundance data.

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6. Conclusions and perspectives

Educating people about the importance of preserving biodiversity as well as to maintain agricultural activities and/or extensive grazing practices (especially in mountainous areas), exerts a prominent role to foster ecosystems' stability and the maintenance of traditional and cultural heritages. Nevertheless, to obtain a positive feedback and reliable results, it is necessary to discern the main factors that may exacerbate the degree of human–wildlife negative interactions with the final purpose to implement adequate mitigation interventions, mainly towards the most conflictual species (real or perceived ones).

The findings presented in the first manuscript revealed that rural inhabitants and hunters exhibited the most negative attitude towards brown bears and grey wolves compared to urban inhabitants and conservationists, whose attitude was more positive. We showed that direct experience with predators as a consequence of the ongoing re–colonization may have affected the degree of acceptance of certain categories and that the long–term coexistence between humans and carnivores does not necessarily imply increased tolerance. To encourage coexistence, we recommend monitoring changes in attitudes over time relative to carnivore population dynamics.

The findings presented in the second manuscript showed that an average minimum number of 1803 crested porcupines (SD = 26.89, CI 95% = 1750–1855) were estimated within an area covering about 17,111 km^2. Since the crested porcupine is considered as "potentially problematic" because of damages to croplands and riverbanks, assessing its abundance is even more important to delineate adequate conservation and management actions to limit the potential trade–off effects over human activities. On a local scale, capture and removals represent only a temporary solution to reduce crop damages since the area is rapidly re–occupied by other individuals. Moreover, the use of visual deterrents and/or electric fences seems to be poorly effective since crested porcupines get rapidly used to these devices and present hollow quills, which make them immune to electrical shocks. To date, tin fences (partly buried) of about 50 cm heigh from the ground seem to be the most effective method to prevent crop damages.

The findings presented in the third manuscript revealed that the different feeding behaviour of brown bears and grey wolves, could lead the latter to be far more problematic than the firsts in terms of attacks towards livestock. In fact, no ‘hotspot’ conflictive areas were observed as far as bear predations are concerned. Conversely, for what concerns wolf predations, a ‘hotspot’ conflictive area was observed in the ‘Lessinia’ highland, where at least four wolf packs are presents and livestock was frequently left free to graze in open pastures, unattended and unprotected during both day and night. Appropriate support should be given by conservationists and carnivore–policy makers to livestock owners to foster the coexistence between large carnivores and humans in the Italian Alps.

The relevance of this research may be seen in the form of using the results obtained to delineate efficient education strategies to increase the tolerance of certain stakeholder categories towards wildlife (especially, large carnivores), and to implement effective mitigation, conservation and management interventions to promote the long–lasting coexistence between wildlife and humans. This study highlighted that, at least for what concerns the official claims, data are mainly collected only for administrative purposes, and therefore are lacking of several information which could considerably contribute to strengthen the analyses and in turn the results achieved (e.g., number and kin of livestock bred in each farm, presence/absence of prevention measures, proper distinction of the area in which the attack occurred, etc.). We are aware that personal information cannot be given due privacy violation issues. However, because livestock attacks most likely frequently and repeatedly occur in the same mountain farms, providing the geographic coordinates and a code identifying each farm (without giving any other personal information) would be of great help to properly assess the magnitude of conflicts. We do believe that the synergistic collaboration between researchers, wildlife technicians and carnivore–policy makers to develop a proper sampling design to collect data in the field would greatly contribute to delineate the most effective management and conservation strategies.

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