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INFORMATION

From: To:	General Secretariat of the Council Working Party on the Environment
N° prev. doc.:	ST 15686/25
Subject:	Request submitted by Gesellschaft zur Schutz der Wölfe e.V for internal review under Article 10 of Regulation (EC) 1367/2006 on the Directive (EU) 2025/1237 of the European Parliament and of the Council, of 17 June 2025, amending Council Directive 92/43/EEC as regards the protection status of wolves – Annexes 31-39

Delegations will find attached Annexes 31-39 to the request on the above-mentioned subject, as received from Gesellschaft zur Schutz der Wölfe e.V [Society for the Protection of Wolves e.V.].

01.04.2025

This review is made with the scope to contributing to the current debate about wolf conservation in Europe.

Title: Uncertain and outdated data should not be used to push for the downgrading of European wolf (*Canis lupus*) populations. Some comments on Blanco & Sundseth (2023) [1]report for the European Commission.

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Introduction

A report submitted to the European Commission by Blanco and Sundseth in December 2023, was used as a scientific benchmark to propose the downgrading of the wolf (*Canis lupus*) from a "specially protected species" (Appendix II) to a "protected species" (Appendix III) in the Bern Convention. This revision could also require the downgrading of the wolf in the Habitats Directive from Annex IV (species requiring strict protection) to Annex V (species whose collection in nature and exploitation could be subject to management measures), thus making the wolf a *de facto* huntable species. Blanco and Sundseth report is a summary of currently available wolf population data, which are, however, partial and imprecise, and should not be used to call for the downgrading. The report, in any case, does not recommend any revision of the protection status of the wolf. Here we offer a review of the Blanco and Sundseth report as a contribution to a critical evaluation of the downgrading proposal, which, as we believe, cannot be based on any scientific evidence.

Comments on Blanco & Sundseth (2023) report for the European Commission.

Upon completion of the downgrading procedures from the status of a strictly protected species (Article 19), under the Annex V of the Habitat Directive wolves could be killed without the need for derogations if the population's "favourable conservation status" (FCS) is not affected (Article 14). FCS cannot be established by decree independently of the best available scientific

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[1] N2K Group EEIG, Blanco, J. C. and Sundseth, K., *The situation of the wolf (canis lupus) in the European Union – An in-depth analysis*, 2023. Link: https://op.europa.eu/en/publication-detail/-/publication/5d017e4e-9efc-11ee-b164-01aa75ed71a1/language-en

data, which, however, are not currently available. The downgrading proposal filed by the European Commission on 7 March 2025 was explicitly based on the Blanco & Sundseth report, December 2023 (BS23), commissioned by the European Commission itself. The BS23 report is mainly a desk review of existing data published by Boitani et al. (2022), an LCIE report that had previously been submitted to the Bern Convention. Additional information was submitted by scientists and experts to the European Commission by 22 September 2023, or collected from regional and national authorities, official web pages; scientific and technical literature and "*consulting national* experts". The Commission explained that this data collection exercise was not a public consultation but a targeted request for input from relevant stakeholders. However, of the 19,000 emails received in response to the call, over 98% wanted to express an opinion on the subject, rather than submit data. In response to a complaint by ClientEarth against the Commission, the European Ombudsman (EO24) opened an inquiry into how the Commission carried out what it referred to as "a targeted data collection" on the impact of the wolf population in the EU. The reliability of the published data and new evidence on wolf population abundance and distributions is doubtful. The BS23 report has not been peer reviewed, does not express any justification for the downgrading, and does not indicate downgrading as a valid solution to improve coexistence.

The core of the BC23 report is show in Table 2.4.1, which presents "*the latest information on wolf population in EU Member States*". As stated by BC23: "*On 17 November 2023, the information gathered from the above was sent to the Ministries of Environment of the Member States for review*". Thus, it seems that the final assessment/approval of the dataset was political rather than be expertly assessed. The data in Table 2.4.1 are mainly rough numbers, not reliable estimates. A statistical estimate must be based on reliable quantitative data and must be expressed in terms of mean and standard error. The error indicates the uncertainty of the estimate and is necessary to evaluate its precision and reliability. In Table 2.4.1 only four out of 27 countries (Finland, France, Italy and Sweden) show some kind of statistical estimates. In Spain there were ">2100" wolves shown, but how many more are there than 2100? The evaluation of wolf trends was entirely based on "*information provided by the Member States competent authorities*". As correctly stated by BS23: "*It is important to note however that there is no consistent or common approach towards assessing trends in wolf populations across the EU. Across the European Union, it is difficult to establish the overall trends*". In the BS23 report, trends are expressed as guesses in purely qualitative terms: increasing, stable or decreasing, which are not informative enough to evaluate the Favourable Reference Values of the wolf populations.

All the information in Table 2.4.1 is biased by the heterogeneity of the source data, because:

- trends may change depending on the longer/shorter period of the surveys;
- population sizes depend on population delimitation, and "*double counting of transboundary packs has not been corrected*" (BC23);
- assessments of population size and distribution procedures are not standardised;
- some countries estimate the number of wolves; others count the number of reproductive pairs and packs; conversion factors from packs to individuals are difficult Boitani et al. (2022);
- wolf numbers change during the annual biological cycles; wolf numbers in late winter might be half or the less wolf numbers in late spring;
- in hunted populations, population size may vary before and at the end of the hunting season;
- data from countries are not comparable and not directly summable without appropriate quality controls.

Table in S2 Appendix in Di Bernardi *et al*. (2025; attached), explains some of the causes of heterogeneity of the source data. Estimates of proportion of wolf range/population size were obtained by variable combinations of 24 different monitoring methods used in 34 European countries. Some countries qualified their estimates just as > or <. Some of them (11/34) qualified exclusively or almost exclusively with > or <: i.e., Albania; Austria; Bulgaria; Denmark; Latvia; Netherlands; Norway; Portugal; Serbia; Sweden; Ukraine. Thus, it is not clear how much of the wolf range and population size has been really covered by each monitoring method. Obviously, the reliability itself of the different methods is highly variable. For instance, information from hunter observations (used in >75% estimates in Romania) and questionnaires (covering the major portion of wolf range/population estimates in Albania, Bosnia, Montenegro, Slovak, Slovenia, Ukraine), are not reliable enough. Some of these countries are the most active in asking for downgrading (Latvia and the other Baltic countries), while they were inputting the less reliable data.

Those fuzzy estimates are in sharp contrast with heavy hunting plans already active or proposed by some EU countries. Latvia, for instance, plans to harvest "*about 50–60%*" of the wolf population annually, including pups and pregnant females, according to the government's Action Plan for Grey Wolf *Canis lupus* Conservation and Management. So, fuzzy estimates are already used to implement destructive wolf removal plans.

We have no doubts that the number of wolves in the EU is increasing. However, we do not believe that the reported increase rates have been based on reliable and verified empirical data:

- 11,193 wolves estimated by LCIE in 2012;
- 19,400 wolves estimated by Boitani et al. (2022);
- 20,300 wolves have been estimated in 2023 by BS23;
- 21,500 individuals by 2022 (Di Bernardi et al. 2025)

Clearly, there has been an increase in wolf population over the decade since 2012. However, while the estimate by Di Bernardi et al. 2025 is only slightly larger than the total number summed up by BS23, they are only marginally different from the total estimate of Boitani et al. (2022) and which the EC accepted when it chose not to support downlisting of wolves under the Bern Convention in 2022. Categorically, this marginal increase in wolf numbers was insufficient to warrant downlisting in 2024. Moreover, the BC23 report itself provides contrary evidence, as it points out based on already known data that "*half of the wolf populations observed in Europe (19) are in an unfavourable conservation status (with 16 unfavourable-inadequate (U1) and three unfavourable-bad (U2)), and that … the conservation status of the wolf under the Habitats Directive is not uniform across the EU*".

Despite the obvious approximations and discrepancies in these data sources, and despite a European Parliament resolution (EP24) recommending the EU Commission to ensure that "*Member States use appropriate monitoring methods for each of the different large carnivore species to allow for the compilation of high-quality, comparable and standardised data for an effective assessment of population levels*", the data assembled by BC23 has been used as a benchmark by the European Commission to push the Standing Committee of the Berne Convention to approve the downgrading of the wolf.

However, the BC23 report fails to highlight clearly enough the weakness and unreliability of the available data, which are useful to implement the necessary improvements in monitoring methods, but that should not be used to support any review of the wolf's protection status. According to the ECJ Council Decision 2022/2489 "*Based on current data, lowering the protection status of all wolf populations is not justified from a scientific and conservation point of view*". Moreover: *protected species that have achieved a favourable conservation status "must be protected against any deterioration of that status*" (Case C-601/22, *WWF Österreich*, para. 44, and Cases C-473/19 and C 474/19, *Föreningen Skydda Skogen*, para. 65 and 66).

Claims by the European Commission that that downlisting would address socio-economic conflicts, like livestock damage, lack robust evidence - as is evidenced by the collation of data on wolf damage on livestock in the EU member states in Table 3.3.1 of BC23. As it is, BC23 admits that sheep depredation by wolves represents an annual killing of only 0.065%, and that in some of the German federal states with the highest number of wolves, the frequency of wolf attacks on livestock has decreased significantly in recent years, which was associated with the use of adequate preventive measures. BC23 note that depredation levels are lower in areas where wolves have never disappeared.

BC23 also note that the existing rules on derogations make it possible to balance different interests against the conservation aims of the Directive. As many have argued, this effectively allows Member States to take action to address specific challenges to livestock - such as bold wolves or susceptible geographical locations or practices – by using derogations of the strict protection regime of Annex IV and thus without the need for downlisting protection. Moreover, a recent study interviewing farmers in Northern Greece identified that wolves frequently became scapegoats for deeper rooted issues such as economic disadvantages, policy deficiencies, and rural depopulation (Petridou & Kati, 2025). Farmers who implemented more effective preventive measures had a lower perception of wolves as a major problem.

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the S2 Appendix. Estimates of the percentage of wolf range / population covered by each monitoring method for 34 European countries.

Di Bernardi C, Chapron G, Kaczensky P *et al*. (2025). S2 Appendix. Estimates of the percentage of wolf range / population covered by each monitoring method for 34 European countries

Country	Camera traps: Min number of individuals	Camera traps: Detection of reproductive units	Camera traps: CMR estimate of population size	Snow tracking: Min number of individuals	Snow tracking: Detection of reproductive units	Snow tracking: Track count index	Non-invasive genetics: Min number of individuals	Non-invasive genetics: CMR estimate of population size	Observations: Detection of reproductive units (observation of pups)	Observations: Hunter observation index	Scat surveys: Identify rendezvous sites	Scat surveys: Confirm presence/scat density index	Howling surveys: Confirm reproduction	Howling surveys: Confirm presence	Damage statistics: Confirm presence	Damage statistics: Damage statistics index	Hunting bag: population reconstruction based on age/sex structure of harvest	Questionnaire: Presence info for a region	Questionnaire: Unconfirmed estimates for region	Opport. presence sign collection: Presence info at different levels of certainty	Opport- presence sign collection: Public observation index	Habitat suitability modelling & density extrapolation	Radio-telemetry to estimate average range size or movement distances	Expert «guestimate»
Albania	<10																	50-75		<10		>75		>75
Austria		<10					>75		<10		<10				>75					>75				
Belgium	>75	>75					>75		>75															
Bosnia and H.	10-25	10-25	Unkn	<10	Unkn	Unkn	Unkn	Unkn	Unkn		<10	Unkn	<10	<10	25-50	Unkn	<10	25-50		Unkn	Unkn	Unkn	Unkn	<10
Bulgaria	<10	Unkn		Unkn	<10	Unkn	Unkn	Unkn	Unkn	>75	<10	<10	<10	<10	<10	Unkn	>75	Unkn	Unkn	10-25	<10	>75	Unkn	>75
Croatia		50-75		25-50	25-50	25-50	25-50	25-50	25-50		<10	10-25			>75	>75		Unkn		50-75		>75	25-50	
Czech Rep.		>75			>75		50-75		<10			>75	25-50	25-50						>75				
Denmark	>75	>75					>75		>75											>75				
Estonia		10-25			>75	>75			>75		10-25		10-25		>75		50-75						<10	
Finland		10-25		25-50	25-50	>75	>75		>75			25-50			<10		>75			>75			10-25	
France	>75	>75	>75	50-75	50-75	50-75	>75	>75	<10				>75		>75					>75	>75			
Germany	>75	>75					50-75		10-25			>75			>75					>75			<10	
Greece	10-25	10-25		<10							<10	<10	10-25	<10	>75			10-25	10-25	10-25			<10	
Hungary	>75	10-25		>75			>75		Unkn	Unkn	Unkn	Unkn	Unkn	Unkn				<10	<10	10-25	<10			50-75
Italy	10-25			10-25			25-50	>75				>75	10-25							>75		10-25		
Kosovo																								25-50
Latvia				<10												<10	>75							>75
Lithuania																	>75			10-25				
Luxembourg															>75	10-25				>75				
Montenegro				25-50						10-25					<10				>75					
Netherlands	Unkn	Unkn	Unkn				Unkn	Unkn	>75	Unkn	Unkn	>75	Unkn	Unkn	>75	Unkn		Unkn	Unkn	>75	Unkn			Unkn
North Mac.																						>75		
Norway		>75		>75	>75		>75	>75	>75						>75					>75				
Poland							10-25	10-25							>75	>75						>75		
Portugal	>75	>75					10-25		>75		>75	>75	>75	>75	>75									
Romania	50-75	50-75	50-75	50-75	>75	>75	25-50	25-50	<10	>75		<10		>75	>75	>75						>75	<10	
Serbia	<10	<10			<10				<10						>75		>75	<10	<10	>75	>75		<10	>75
Slovak Rep.				10-25	10-25		10-25	10-25										>75	>75					
Slovenia		Unkn					>75	>75	Unkn		10-25		>75	>75	>75			>75					<10	
Spain		10-25							25-50				50-75										<10	<10
Sweden					>75			>75	>75															
Switzerland		10-25			<10		>75		10-25		10-25		25-50	10-25	>75	>75				>75				
Türkiye																						>75		10-25
Ukraine		<10		>75	<10							<10					<10	>75						

this review is made available for public dissemination

Large carnivore distribution maps and population updates 2017 – 2022/23

Version 1.2 – with updated population estimates

JUNE 2024

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Large carnivore distribution maps and population updates 2017 – 2022/23

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Contents

1. Background

Large carnivores have made a remarkable comeback in Europe during the last half century, and recovery is still ongoing in large parts of the continent (Andrén 2018; Boitani 2018; Boitani et al. 2022; Chapron et al. 2014; Huber 2018; Ranc 2018; von Arx 2020). While this expansion can be celebrated as a huge conservation success, it also creates considerable challenges for coexistence in the multi-use landscapes of Europe (Linnell 2013).

Having a common understanding of the distribution, size and trends of large carnivore populations in Europe is one prerequisite for a knowledge-based dialogue in the often heated and highly politicized discussions about future scenarios of large carnivore conservation and management in Europe. Because of the scale at which large carnivores utilize the landscape it is essential to conduct periodic continental scale assessments of their status transcending sub-national and national borders. Assessments at this scale require harmonising diverse datasets that arise from different jurisdictions using different monitoring approaches.

This document provides the best available overview of brown bear (*Ursus arctos*), Eurasian lynx (*Lynx lynx*), wolf (*Canis lupus*), golden jackal (*Canis aureus*), and wolverine (*Gulo gulo*) distributions and population sizes at a continental scale. The data in this report is based on over 200 national experts who are co-authoring the report (Appendix 1) plus a huge number of additional regional and local collaborators and supporting agencies (Appendix 2).

References listed in overview tables are not part of part IV Literature but can be found in the compilation of the most recent publications on population and rage estimates in Appendix 6.

2. Methods

2.1. Distribution mapping methods

The mapping approach generally follows the methods described in (Chapron et al. 2014) and (Kaczensky et al. 2013). It updates the published Species Online Layers 2012-2016 for brown bear, Eurasian lynx, wolf, golden jackal, and wolverine (Kaczensky et al. 2021; Ranc et al. 2022) for the period 2017-2022/23.

Large carnivore presence was mapped at a 10 x 10 km (ETRS89-LAEA Europe) grid scale. This grid is widely used for Habitat Directive reporting to the European Union (EU) and can be downloaded at: http://www.eea.europa.eu/data-and-maps/data/eea-reference-grids-2. The map encompasses the continental EU countries plus Switzerland and Norway, and the EU candidate / potential candidate countries in the Balkan region, in addition to Ukraine and Turkey. For the two latter countries, only parts were included; for Ukraine only the Carpathian region (for this report Ukraine was artificially cut off and the straight line in the east does not represent the national border), and the European part of Turkey (Fig. 1).

For the 2012-2016 mapping, several countries were not or not fully (not for all species) included (Hungary, Montenegro, Turkey), so that no comparisons can be made of the updated carnivore distributions with those from the last mapping for these countries.

Mapping large carnivores for this report had a two-fold goal:

- Visualizing areas of large carnivore presence
- Visualizing the variation in the underlying data quality

Fig. 1: Spatial extent of the large carnivore mapping area and the focal species: brown bear (Ursus arctos), Eurasian lynx (Lynx lynx), wolf (Canis lupus), golden jackal (Canis aureus), and wolverine (Gulo gulo). Note: Ukraine was cut to only include the Carpathian region and only the European part of Turkey was included for the mapping.

2.1.1. Presence status

We aimed to distinguish between two presence levels:

- **Permanent** = suggesting an established population which is reproducing, but also including cells with continuous presence in the absence of documented reproduction.
- **Sporadic** = suggesting only occasional presences of dispersers or lone individuals.
- Where this distinction was not possible, but presence was confirmed, we used **Undefined** = presence confirmed, but not known if it is permanent or sporadic.

"Permanent" is equivalent to the status of "Present regularly" (PRE) as used in Article 17 reporting to the Habitats Directive, while "Sporadic" corresponds to the status of "Occasional" (OCC) in the same system. It was not possible to systematically separate out the "Newly arrived" (ARR) category, although it may well apply to the many new jackal records throughout western and northern Europe. Finding a common harmonized definition that fits all monitoring circumstances is difficult and the distinction required expert assessment. Here are the most common scenarios that we have continued with from the previous 2012-2016 mapping cycles:

1) For **countries where the known annual species distribution was monitored annually**, the distinction between permanent and sporadic was primarily made based on how consistently the species was detected in a cell over the 5–7-year monitoring period:

- **Permanent** = presence confirmed in ≥ 3 years in the last 5 - 7 years OR reproduction confirmed at least once within the last 3 years
- **Sporadic (highly fluctuating presence)** (presence confirmed in <3 years in the last 5 years OR in <50% of the time)

2) For **countries where the probability of species presence is modelled** based on presence signs in combination with habitat parameters and distance rules, the distinction between permanent and sporadic can be made based on the modelled "probability of presence" value of a cell. As models used for different populations will vary in their approach, **the cut-off values for permanent, sporadic, and absent were defined by the national/population level species experts**.

3) For **countries where the total range is covered by rotating annual surveys of parts of the total area over a 5–7-year period** (i.e. different sections are surveyed in different years such that the whole area is surveyed at least once during the cycle), other criteria need to be used such as: comparison to presence in a cell (or adjacent cells) during the previous survey cycle, or confirmed reproduction, or the presence of females, to delineate permanent presence from sporadic presence. However, where monitoring is too fragmented and infrequent so that no reasonable distinction between permanent and sporadic can be made, the category "undefined" was used.

In general, telemetry data of long-distance dispersers out of the known range and once off documentation of individuals outside the known range were categorised as sporadic presence.

2.1.2. Large carnivore signs used to map presence

Large carnivore signs as a basis for mapping

We used the following presence categories, which were derived from the SCALP criteria for lynx in the Alps (Molinari-Jobin et al. 2012), but supplemented with two additional data quality information categories:

1. Confirmed presence signs

- **Category 1 (C1)**: "Hard facts", verified and unchallenged large carnivore presence signs (e.g. dead animals, DNA, verified camera trap images);
- **Category 2 (C2)**: "Confirmed signs", large carnivore presence signs controlled and confirmed by a large carnivore expert (e.g. trained member of the network), which requires documentation of large carnivore signs (e.g. tracks in the snow).
- We also had to include the category C1*, referring to C1 records which also include observations by trained or experienced personnel; it was not always possible to know if these observations also included non-documented records such as direct observations (which by the SCALP definitions do not quality as confirmed records). This is particularly true for the documentation of bear family groups or where monitoring was heavily based on hunters, foresters, and protected area wardens (e.g., Croatia, Slovakia, Ukraine).

2. Extrapolated confirmed presence signs

- **Category "buffered":** Confirmed presence signs with a buffer around them, ideally based on well documented/ published methods, and usually done to represent home ranges that are typically larger than 10 x 10 km in many parts of Europe (especially in the north).
- **Category "modelled": confirmed presence signs and modelling** based on habitat suitability and/or proximity criteria ideally based on well documented/published methods and explicit cut-off values.

For areas of poor monitoring coverage or infrequent monitoring, we also included:

3. Unconfirmed presence signs

- **Category 3 (C3)**: Unconfirmed reports of category 2 large carnivore presence signs and all presence signs such as sightings and calls which, if not additionally documented, cannot be verified.
- **Category "Soft":** Extrapolation of large carnivore presence based on interviews questionnaires, and media coverage from 2017-2022/23,
- **Category "Past presence":** Documented presence from the past (but no older than from 2010) and no indication that the situation has changed.

From signs to grid cell

Ideally, GPS locations of large carnivore signs were intersected with the 10 x 10 km ETRS89-LAEA Europe grid. However, for some countries, data was collected at the spatial scale of hunting grounds (e.g., bear observations by hunters in eastern and south-eastern Europe). In this case the hunting grounds with large carnivore presence were intersected with the 10 x 10 km grid. Large carnivore presence was assumed for all cells intersecting the hunting ground (normally using a minimum intersection area in the range of >10%).

2.1.3. Data quality of large carnivore presence cells

We aimed to present data quality information at the grid cell level, but for some datasets it was not feasible to do so, and data quality was provided at the scale of the entire, or parts of, the layer, and not the individual grid cells. For some datasets this resulted in a mix of cells based on confirmed and buffered, modelled or unconfirmed cells without spatially explicit information at the grid cell level (Fig. 2).

Consequently, the following final data quality categories were used:

- **Confirmed presence:** based on C1 & C2 signs, cases where C1 included unspecified "observations" were marked with a star (C1*)
- **Extrapolated presence:** cells which don't have LC signs but are intersected by buffers or have a high probability of large carnivore presence based on documented modelling approaches.
- **Unconfirmed presence:** cells with only C3 signs, or cells with only data from prior to 2017, where presence is still assumed to persist.

Where data was only available at the shape file level, we also used the following categories:

- **Confirmed and extrapolated presence:** mixed layer with buffered C1 & C2 signs and/or documented modelling approach, or when data was only available at the level of hunting grounds.
- **Confirmed and unconfirmed presence:** a mixed layer with C1-C3 signs; for these datasets it can be assumed that the majority are C1 and C2, but that documentation is not (readily) available - these data sources include hunter observations and some damage inspection data.

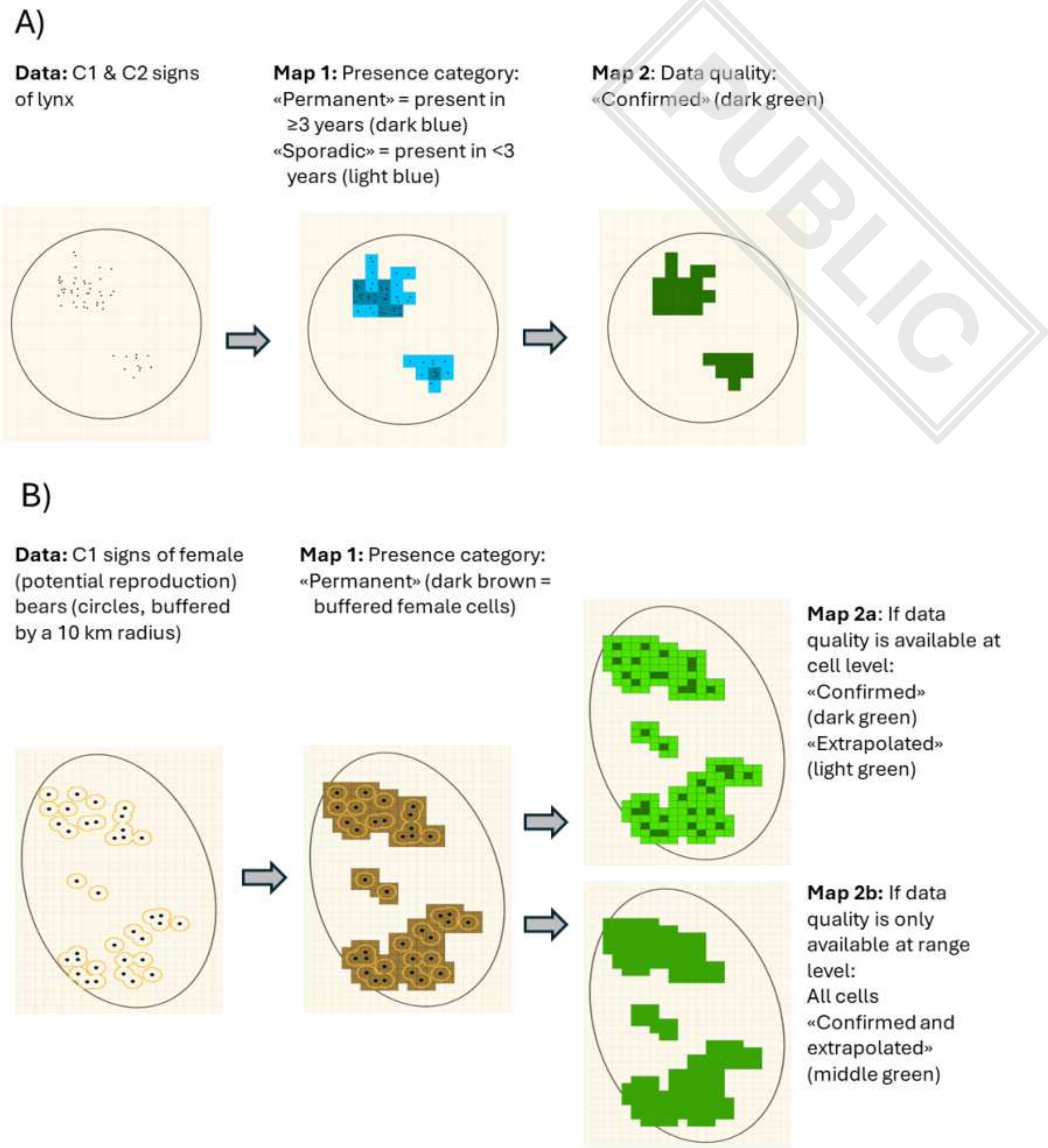

Fig. 2: Conceptual figure showing how presence and data quality were derived based on different datasets and the level at which information was available.

A) Dataset where presence over time was used to distinguish between permanent and sporadic presence in lynx

B) Dataset where presence signs were collected sequentially and where the presence of females (=potential for reproduction) with a 10 km buffer (orange ring) was used to distinguish between permanent and sporadic presence for brown bears in Scandinavia in the cumulative map. Data quality (confirmed signs versus extrapolated) was available at the cell level for map 2a and at the shape file level for map 2b.

2.1.4. Populations

Grid cells were assigned to populations based on the LCIE's population approach (Linnell et al. 2008). Population delineation follows a combination of ecosystem boundaries, topography, different management regimes, distributional discontinuities and administrative units, selected to create practical and functional management units. Population borders were partly drawn out of convenience, roughly following topographic regions, natural or artificial barriers such as large rivers, and in some cases national borders to facilitate reporting. Ecological conditions and monitoring methods tend to be similar within populations.

Populations are primarily based on where animals are detected, not where they have originated from (except for reintroduced populations). Hence even if the genetic origin of an individual was known (e.g., based on genetic analysis), the cell it shows up in will not represent the animal's origin, but rather the population of the location at which it was detected. Cells in-between existing populations and geographic regions and single cells outside of existing populations were given the Status "Unassigned" - even if their genetic origin was known. An exception to this concerns lynx which derive from many reintroductions. Here we have used the origins as an additional factor in population separation.

In the last cycle of large carnivore mapping for 2021-2016, we delineated 11 populations for lynx, 10 for bears, 9 for wolves, 4 for golden jackals, and 2 for wolverines (Kaczensky et al. 2021, Ranc et al. 2022; Fig. 3).

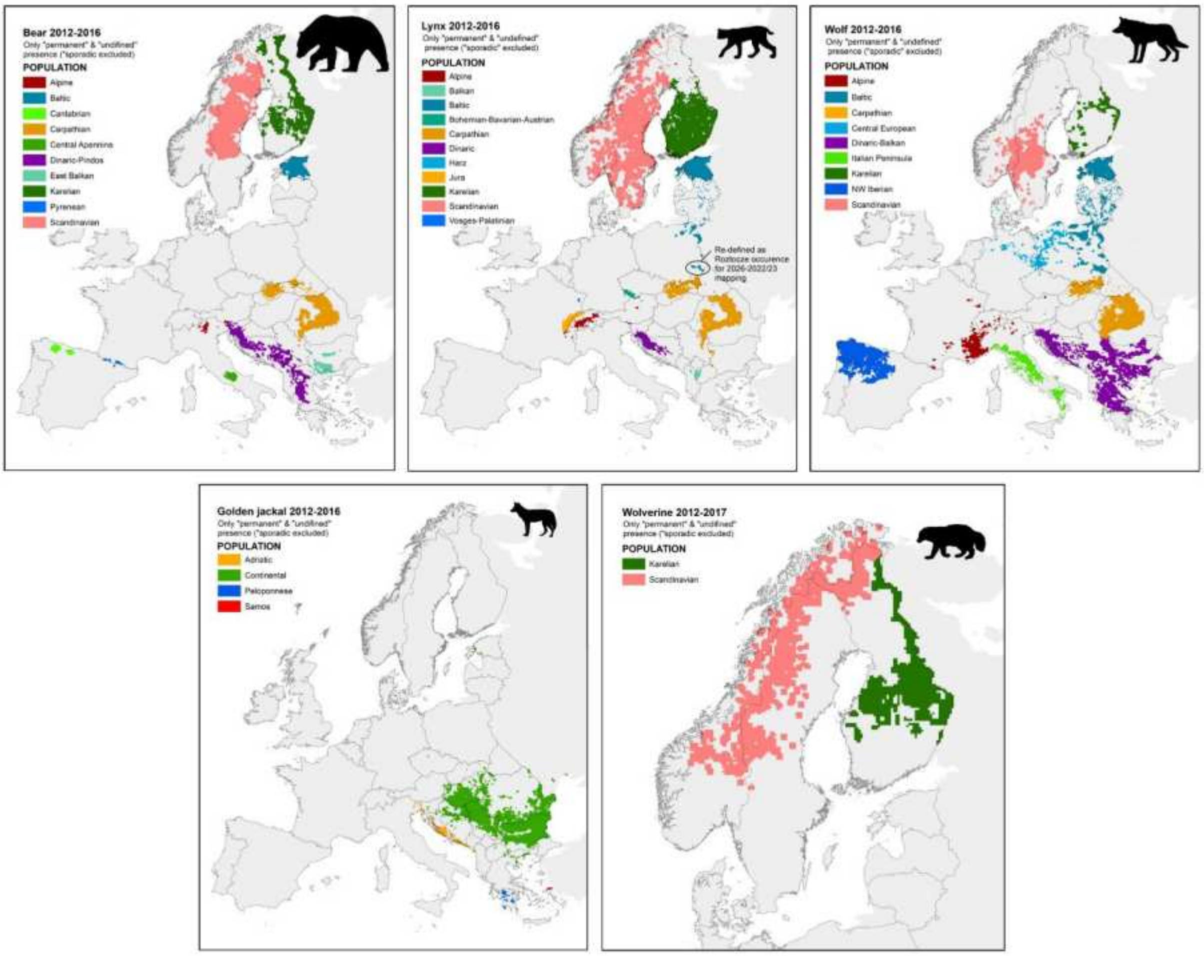

Fig. 3: Population delineations for large carnivores in Europe for the period 2012-2017.

We generally retained the same population delimitations but, made a few small adjustments to represent the changing situation on the ground (expansion and re-connection of populations) and new scientific information:

- **For wolves**, we changed the name of the “Central European Lowland” wolf population to “Central European” and the name of the “Northwest Iberian” wolf population to “Iberian”, reflecting the spread of theses populations into mountain ranges and towards the east, respectively. For wolves in Germany, we used the shapefile of the perimeter of the Alpine Convention to delineate between “Central European” and “Alpine” wolf populations. For the Polish part, we used the delineation shown in ((Szewczyk et al. 2021) Fig. 1A). The “Alpine” population now also includes large adjacent areas in Italy and France (Fig. 10 & 11).
- **For lynx**, we separated the reintroduced population in northwest Poland from the Baltic population given its recent and different genetic origin and named it the “Pomeranian occurrence” in accordance with lynx population subdivision by the Cat Specialist Group (U. Breitenmoser pers. com. 2024). One other recent re-introduction project has resulted in another new occurrence: the “Black Forest – Swabian Jura occurrence” in southwestern Germany.
- **For wolverine**, we now drew a straight line from the Norwegian-Finnish-Russian border in the north to the southernmost point of the Finnish-Swedish border to delineate the Scandinavian from the Karelian population, which is convenient and in line with recent genetic results (Lansink et al. 2020; Lansink et al. 2022).
- **For Slovenia** – due to a request from national authorities - we used the national border of Slovenia in the north to delineate **bears** and **wolves** between the Alpine and Dinaric-Pindos / Dinaric-Balkan populations. Therefore, all wolves and bears on the territory of Slovenia are assigned to Dinaric population (where the majority likely originates from the Dinaric population). However, for delineating **lynx**, the geographic separation between the Alps and Dinaric Mountains (roughly marked by the highway A1) was used as the lynx in the Slovenian Alps still seem to be functionally separated from the Dinaric lynx (M. Krofel pers. comm. 2024).

2.1.5. Border cells

- Many populations occur along and across national borders and cells are shared by neighbouring countries. Only a few populations reported in a coordinated way for a cross-border region (e.g., for bears in the Pyrenees, lynx in the Alps, and all species in Norway & Sweden). Where large carnivore presence was detected in the same cell by neighboring countries, the cell with the “better information status” wins, following the rule:
    - Permanent > Sporadic > Undefined
    - Confirmed presence > Confirmed and extrapolated presence > Confirmed and unconfirmed presence > Extrapolated presence > Unconfirmed presence

2.1.6. Time period

The majority of the large carnivore presence layers cover the time period 2017-2022/23. But because monitoring conditions and logistics vary between countries, regions, and species, we documented the specific monitoring periods. Periods referring to 2022/23 usually mean that the species range or population estimates are reported at the end of winter in early 2023. However, the use of 2022 for the calendar year versus 2022/23 for the biological/monitoring year was not consistently used and therefore 2022/23 can mean: only the first part of 2023 is included, all of 2023 is included, or all or parts of 2023 and the first part of 2024 is included. For all countries that provided data deviating from the 2017 start and 2022 or 2023 end date, we explicitly label the monitoring period on one map together with the range.

2.1.7. Change in range since the last mapping for 2012-2016

We visualised the gains and losses of cells with large carnivore presence of any presence category (undefined, permanent, sporadic pooled) between the two periods 2012-2016 and 2017-2022/23. We additionally compared the cell count between the two time periods by country and population.

The latter were critically reviewed to establish whether these gains or losses reflect true range changes or are just the result of altered monitoring methods or changes in monitoring effort. The latter information was obtained through an online questionnaire survey (Appendix 4), which asked about mapping details, assessment of the trend in distribution, changes in monitoring methods and effort, and the main monitoring method. We also took into account that certain countries were not included in the mapping exercise for 2012-2016.

2.1.8. Shape file information

Together with this report, we also provide the species distribution shape files (one file for each species) for 2017-2022/23. The shapefiles include the following metadata:

Metadata table	**Information provided**
FID	Unique identifier ID
CELLCODE	10x10 km ETRS89-LAEA (Lambert Azimuthal Equal Area) Europe grid ID
EOFORIGIN	East coordinate in ETRS89-LAEA projection (EPSG:3035)
NOFORIGIN	North coordinate in ETRS89-LAEA projection (EPSG:3035)
COUNTRY	Country (in some cases large transboundary region)
PERSON	Person(s) who compiled and sent the map
SPECIES	*Canis lupus*, *Canis aureus*, *Gulo gulo*, *Lynx lynx*, or *Ursus arctos*
POPULATION	Species-specific population as defined by LCIE
PRESENCE	Presence category: Undefined, Permanent, or Sporadic
DATAQUAL	Data quality categories: see section 2.1.3. above
DATASOURCE	Short reference of data source – for details see Appendix 6
YEAR	Time period the data layer covers
YRCOMPILED	Year the maps were compiled: 2024
COMPILERS	Kaczensky, Ranc, Hatlauf, Payne *et al.* 2024 for the Large Carnivore Initiative for Europe (LCIE)

2.1.9. Questionnaire survey

Information on mapping and distribution monitoring details were obtained via an online questionnaire consisting of 30 questions asking about mapping details, assessment of trend in species distribution, main range monitoring methods, and most recent supporting references or publications on status and distribution of the target species (Appendix 4).

2.2. Population estimation methods

We obtained population estimates from regional and national experts following the methods described previously (Boitani et al. 2022; Chapron et al. 2014; Kaczensky et al. 2013). In short, we used an online questionnaire (Appendix 5) distributed to members of the LCIE who then distributed it further to their contacts.

The questionnaire included 24 questions asking for population estimates of the target species at national and population segment level, the form of transboundary cooperation, trends, main monitoring methods, management plans, and population goals, as well as references to the most recent population estimates (see Appendix 6 for compilation) and management plans (see Appendix 7 for compilation).

We subsequently compiled an overview of main monitoring methods and population estimates at population level and compared numbers to those obtained by LCIE for 2012-2016 (see: https://www.lcie.org).

3. Results

3.1. Mapping results

Large carnivore distribution maps were compiled for all 24 mainland EU countries (the three island nations don't have large carnivore populations, namely Ireland, Malta, and Cyprus). Additional maps were compiled for another 10 European countries/regions, namely Switzerland, Norway, Bosnia and Herzegovina, Serbia, Kosovo*, Montenegro, Albania, and North Macedonian, plus the Carpathian region of Ukraine and the European part of Turkey because of the continuity of species distributions. The microstates of Andorra, Lichtenstein, San Marino, Monaco, and Vatican City are not explicitly listed due to their small size but are covered via monitoring of the neighbouring countries whose grid cells overlap the microstates.

In total this report provides distribution maps for 34 European countries/regions. This meant that we obtained maps for 3 more countries than for the period 2012-2026, namely Hungary, Montenegro, and the European part of Turkey. The latter needs to be kept in mind when comparing maps from 2012-2016 with those from 2017-2022/23.

Belarus was initially included and colleagues working in Belarus provided us with information on the large carnivore range. However, while presence data from camera traps was available at the 10 x 10 km grid for two study areas for lynx and wolf (Kudrenko et al. 2023; Palmero et al. 2023), the national monitoring is based on a 50 x 50 km of the Atlas of Mammals of Belarus. The data in the mammal atlas suggests widespread presence of wolf and lynx throughout the entire country, a more restricted, but still widespread presence of the brown bear, and some presence of the golden jackal. The huge difference in scale and the different grid projection did not allow for integrating this very coarse-grained data into our mapping framework in any comparable way. In addition, in November 2022 Poland build a 199 km impermeable border fences along the border with Kaliningrad and recently a 180 km impermeable border fence along the border with Belarus. Estonia, Latvia and Lithuania also have border fences towards Belarus and Russia. The sum of these fences making the exchange of large carnivores between Belarus and Russia, and the rest of Europe extremely challenging (Nowak et al. 2024). As a result, we excluded Belarus from our maps.

A note of caution:

The overview tables on mapping and distribution monitoring methods are based primarily on questionnaires with predefined categories that greatly simplifies the diversity of monitoring and mapping approaches. Even if the same general categories were used, other details differed but cannot be reflected here in all detail. In addition, different people interpreted categories differently. Examples of problems that arouse include:

- Main distribution mapping method – it was unclear if it meant the % of the range monitored with the method or the % of data used for the mapping; this was not carefully defined and likely resulted in people answering in different ways
- "% Known range monitored" – it was not clearly defined what is meant by active monitoring versus passive / opportunistic monitoring (e.g., is the information on culled/hunted bears active or passive / opportunistic monitoring? – it can be interpreted both ways).

* This designation is without prejudice to positions on status, and is in line with UNSCR 1244/1999 and the ICJ Opinion on the Kosovo declaration of independence

- There were some discrepancies between information provided in the questionnaire and the GIS data files, which could not always be fully resolved.

In summary, the tables show larger scale patterns, but may not be completely comparable for some countries/regions.

The maps represent the best available data on large carnivore distribution in Europe. However, detecting large carnivores is not only dependent on their presence, but also on the effort spent searching for their signs or collating and verifying reports. Thus, where monitoring is scarce or absent, presence can go undetected, especially if it is only sporadic. The smaller the large carnivore population and the more recent their appearance in a country, the more likely that they are detected and documented because of the increased effort that is used to follow rapidly emerging situations. Where populations are large and well established, on the other hand, there is often no capacity to document every large carnivore which happens to sporadically show up outside the permanent range.

The distinction between "permanent" and "sporadic" presence would be best done based on the presence or absence of confirmed reproduction. We tried to obtain data on reproduction for the 2012-2016 mapping, but it was not feasible at the continental scale – too many countries do not have the monitoring capacity to do so.

The current method to use re-occurrence of large carnivore signs in a 10 x 10 km cell over the monitoring period with or without information on reproduction is a compromise. It works quite well for countries with annual monitoring of the known range but does not allow us to distinguish between areas of an established population and areas of constant reoccurrence without reproduction. This is particularly true for bears which show strongly male biased long-distance dispersal where you can have areas with regular presence of male bears on the dispersal front far from areas with reproduction (Kojola et al. 2003; Swenson et al. 2001).

Consequently, "permanent" presence occurs in parts of the Alps (e.g. Switzerland and the eastern Alps) where no reproduction has yet been recorded. However, in respect to bear presence and perception by local people, bears are permanently present in these areas now.

Where monitoring data accumulates over the monitoring period through sequential surveys of different portions of the total range, re-occurrence cannot be used to distinguish between "permanent" and "sporadic". Where specific data on reproduction is available it is a good substitute, but where it is not available all cells were either given the status of "undefined" presence, or other criteria such as data quality or locality were used to distinguish between "permanent" and "sporadic". While this results in a somewhat subjective expert-based assessment of the presence category, we still believe there is a value in trying to separate between "permanent" and "sporadic" to understand species range dynamics.

In conclusion, changes in the large carnivore distribution range at the European scale between 2012-2016 and 2017-2022/23 must be interpreted with a focus on the "permanent" distribution but also need to consider changes in monitoring methods and effort, and trends in population size.

3.1.1. Brown bear

Overview of main distribution monitoring methods

The most important monitoring methods for determining bear distribution were dead animals (mainly from hunting/culling but also traffic kills), non-invasive genetics (hairs and scats), camera traps, damage statistics (livestock and beehives), observations of females with cubs, and interviews (Table 1). Methods under "other" are hunter observations at feeding stations (structured surveys conducted twice a year in Croatia) and direct observations by hunters (Slovakia).

Table 1: Main distribution monitoring methods for brown bears in Europe.

Country	Range monitoring method												
	Dead animals	Non-invasive genetics	Camera traps	GPS tracking	Active snow tracking	Howling surveys	Family groups	SCALP C2	Damage statistics	SCALP C3	Quest. & interviews	Past presence	Other*
Brown bear													
Albania	<10%	<10%	25-50%	<10%							10-25%	<10%	
Austria	<10%	50-75%	10-25%					<10%			>75%		<10%
Bosnia & Herzegovina	>75%	10-25%	50-75%					10-25%			<10%		
Bulgaria	<10%		25-50%	<10%				50-75%	50-75%	<10%			
Croatia	>75%	>75%	25-50%	10-25%			10-25%	>75%	50-75%		50-75%		>75%
Czech Republic		<10%	50-75%	<10%	10-25%			50-75%					
Estonia	<10%		25-50%				50-75%	<10%	<10%		>75%		
Finland	10-25%						50-75%	50-75%	10-25%		>75%		
France	>75%	>75%	>75%					>75%	>75%				
Germany		>75%	>75%					>75%					
Greece	10-25%	>75%	>75%	10-25%			25-50%	50-75%	>75%	10-25%	25-50%	10-25%	
Hungary		<10%	50-75%		10-25%		<10%	50-75%			>75%		
Italy - Alps	>75%	>75%	>75%	>75%			>75%	>75%	>75%	>75%			
Italy - Apennine	<10%	50-75%	25-50%	50-75%			50-75%	50-75%	50-75%			10-25%	
Kosovo*	no dedicated monitoring - by-catch from lynx camera trap monitoring												
Latvia	<10%	25-50%	25-50%				25-50%	>75%	>75%		>75%		
Lithuania			10-25%					50-75%		10-25%	>75%		
Montenegro		10-25%	10-25%	10-25%	25-50%			50-75%	<10%		<10%	25-50%	
North Macedonia	<10%	10-25%	25-50%		<10%		<10%	10-25%	<10%	<10%	<10%		
Poland	<10%								10-25%	<10%	50-100%		
Romania	<10%		<10%		>75%		<10%	<10%	50-75%	10-25%		>75%	
Serbia	<10%	<10%	50-75%	25-50%			<10%		<10%				
Slovakia	>75%	>75%	>75%	10-25%	>75%		>75%	>75%	>75%	>75%	>75%		>75%
Slovenija	>75%			<10%					<10%		>75%		
Spain	>75%	>75%	>75%	25-50%			25-50%	>75%	>75%				
Sweden	>75%	>75%							>75%				
Norway	>75%	>75%	>75%						>75%				
Switzerland	<10%	25-50%	25-50%					25-50%	<10%				
Ukraine	<10%	<10%	<10%	<10%	10-25%	<10%	<10%	<10%	<10%	25-50%	<10%		

More details on species monitoring methods can be found in the compilation of the most recent literature on population size and distribution estimates in Appendix 6.

Overview of main mapping methods

Distribution data was available for most countries up to 2022/23, with only Norway and Sweden having a one year shorter period (until 2021). Most of the bear presence is based on confirmed (C1 & C2) signs intersected directly with the 10 x 10 km grid.

Out of 29 countries/regions with bear presence, 6 use buffers around bear signs (Albania, Latvia, Montenegro, North Macedonia, Sweden, Norway) with 2 (Albania and North Macedonia) additionally using modelling for extrapolated presence. Three countries (Croatia, Slovakia, Ukraine) have some, or most, information collected at the level of hunting grounds, which were intersected with the 10 x 10 km grid.

For 6 countries (Croatia, Montenegro, Sweden, Norway and in part Slovakia and Ukraine) information on data quality was not or only partly available at the individual cell level (Table 2).

Table 2: Mapping details for brown bear in Europe.

Country / Region	FINAL_time	Spatial scale	% Known range monitored		Large carnivore signs used	Definition of gridcells based on	Scale of data quality information	Presence categorisation based on	Method change	Range trend estimate since 2012-2016	
			Active	Passiv						Trend	Assessment
Brown bear											
Albania	2017 – 2022/23	Only reference areas	15	20	C1 & C2	Buffered confirmed presence signs & modelling overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Austria	2017 – 2022/23	Entire known range	0	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence only	No	Fluctuating	Real
Bosnia & Herzegovina	2017 – 2022/23	Entire known range	60	30	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real & method change
Bulgaria	2017 – 2022/23	Only reference areas	45 - 60	25-30	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Yes	Fluctuating	Real & method change
Croatia	2019 – 2023	Entire known range	100	100	C1* & C2	Hunting grounds with confirmed presence signs overlaid with the 10 x 10 grid	Country level	Re-occurring presence and/or reproduction	No	Increasing	Real
Czech Republic	2017 – 2023	Entire known range	80	20	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Fluctuating	Real
Estonia	2018 – 2022/23	Entire known range	100	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	No obvious change	Real
Finland	2017 – 2022/23	Entire known range	100	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Reproduction only	Yes	Increasing	Real
France	2017 – 2022/23	Entire known range	90	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Germany	2017 – 2022/23	Entire known range	0	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence only	No	No obvious change	Real
Greece	2017 – 2022/23	Entire known range	70-80	20-30	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Hungary	2017 – 2022/23	Entire known range	50	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Italy - Alps	2017 – 2022/23	Entire known range	90	90	C1* & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Italy - Peninsula	2017 – 2022/23	Entire known range	60	90	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Yes	Increasing	Real & method change
Kosovo*	2016 - 2023/24	No information			C1 & C2-C3	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Data quality	Unknown		
Latvia	2017 – 2023	Entire known range	10	100	C1 & C2	Buffered confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Lithuania	2018 – 2023	Entire known range	0	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence	Yes	Increasing	Real
Montenegro	2018 – 2023	Entire known range	60	60	C1 & C2	Buffered confirmed presence signs overlaid with the 10 x 10 grid	Country level	Unknown	No	No obvious change	Real
North Macedonia	2017 – 2022/23	Entire known range	0	60	C1	Buffered confirmed presence signs & modelling overlaid with the 10 x 10 grid	Cell level	Geographic location	No	Unknown	More data needed
Poland	2017 – 2022	Entire known range	15	85	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Yes	No obvious change	Method change
Romania	2017 – 2022/23	Entire known range	70	30	C1, C2 & C3	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Data quality	Yes	No obvious change	Method change
Serbia	2017 – 2022/23	Entire known range	75	25	C1	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Slovakia	2017 – 2022/23	Entire known range	100	100	C1 & C2, C1*-C3	Hunting grounds with confirmed presence overlaid with the 10 x 10 grid & additional C1 & C2	Cell level & country	Re-occurring presence and/or reproduction	No	Increasing	Real
Slovenia	2017 – 2022/23	Entire known range	99	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Yes	No obvious change	Likely expansion
Spain - Cantabrian Mnts	2018 – 2023	Entire known range	80	20	C1* & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence	No	Increasing	Real
Switzerland	2017 – 2023	Entire known range	0	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence only	No	No obvious change	Real
Ukraine - Carpathians	2017 – 2023	Carpathians	10	100	C1*-C3	Hunting grounds with confirmed presence overlaid with the 10 x 10 grid & additional C1 & C2	Cell level & country	Hunter density estimate	Yes	Fluctuating	Real
Sweden	2017 – 2021	Entire known range	100	100	C1 & C2	Buffered confirmed presence signs overlaid with the 10 x 10 grid	Country level	Reproduction only	No	No obvious change	Real
Norway	2017 – 2021	Entire known range	100	100	C1 & C2	Buffered confirmed presence signs overlaid with the 10 x 10 grid	Country level	Reproduction only	No	No obvious change	Real

A distinction between permanent and sporadic presence was primarily made based on re-occurrence and/or reproduction. However, 3 countries (Finland, Sweden, Norway) exclusively used reproduction, or the presence of female bears (=the potential for reproduction), to define the permanent range. A few used data quality (Kosovo*, Romania), geographic location (North Macedonia), or hunter density estimates (Ukraine); for Montenegro the criteria used were unknown.

The trend in bear distribution was estimated as increasing in 14 countries/regions, showing no obvious change for 9, fluctuating for 4, and unknown for 2. No country/region reported a decreasing range (Table 2).

Current brown bear distribution in Europe

Brown bear presence has been documented in 29 of the 34 countries/regions monitored. The species is totally absent from Belgium, Denmark, Luxembourg, the Netherlands, and Turkey. In Portugal, a brown bear was recorded for the first time in a border cell in 2019 (Fig. 4).

The total distribution area occupied by the brown bear in Europe currently covers ca. 1.2 million km^2, which is a 4.6% increase in distribution since 2016 (Table 3).

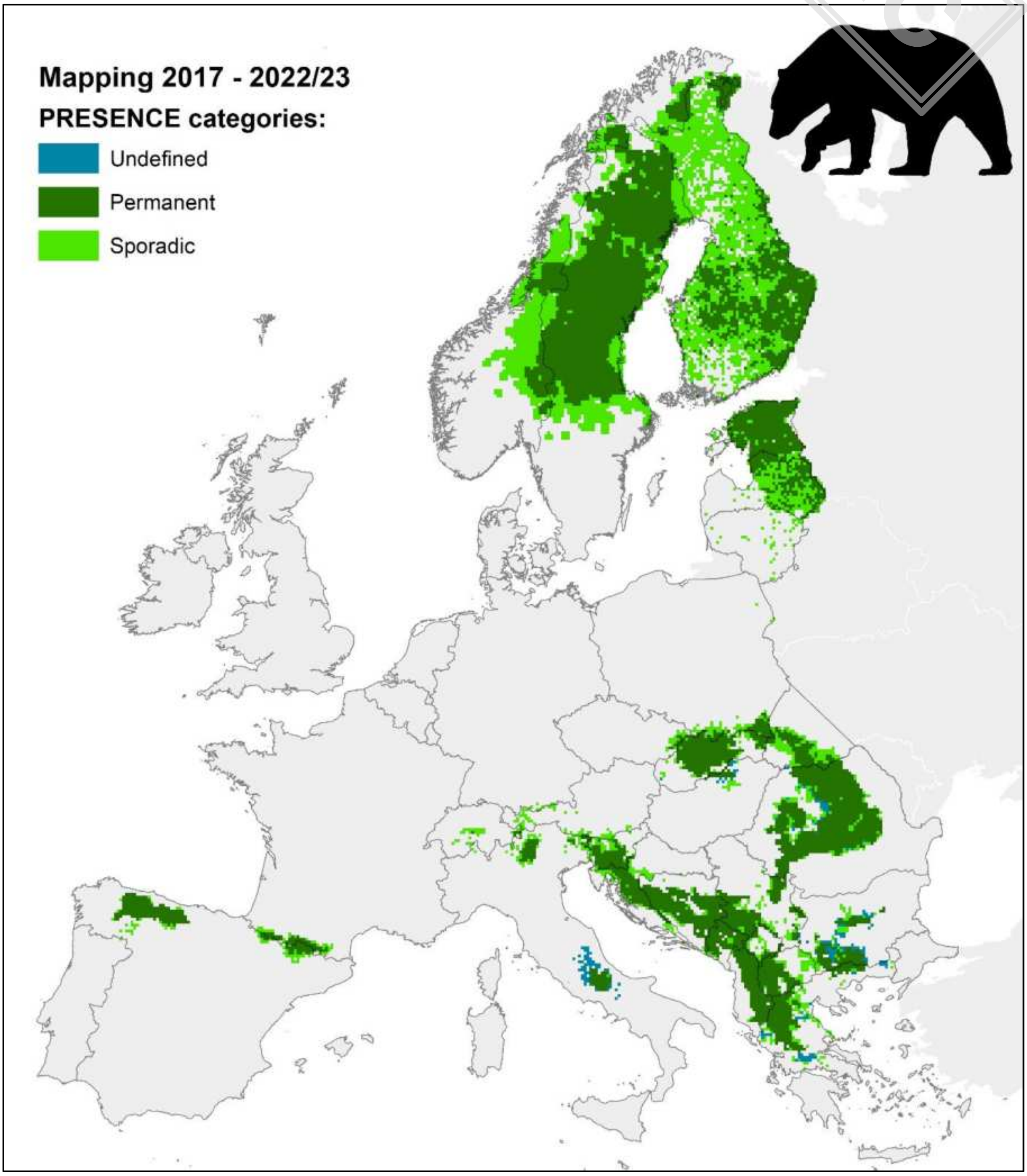

Fig. 4: Brown bear distribution in Europe for the period 2017-2022/23.

Most of the bear distribution in all populations is based on confirmed bear signs, often in combination with some form of extrapolation (buffers and/or modelling, see Table 2). The Dinaric-Pindos population still has the least robust data foundation. In parts of the Carpathian population monitoring is dependent on observations from hunting grounds or protected areas. These observations are often less formally documented or accessible and include direct observations and were given the mixed status of "Confirmed and unconfirmed" data quality. Where available, this data was confirmed with C1 & C2 data from dedicated monitoring projects, especially those using camera trapping or non-invasive genetic monitoring (Fig. 5).

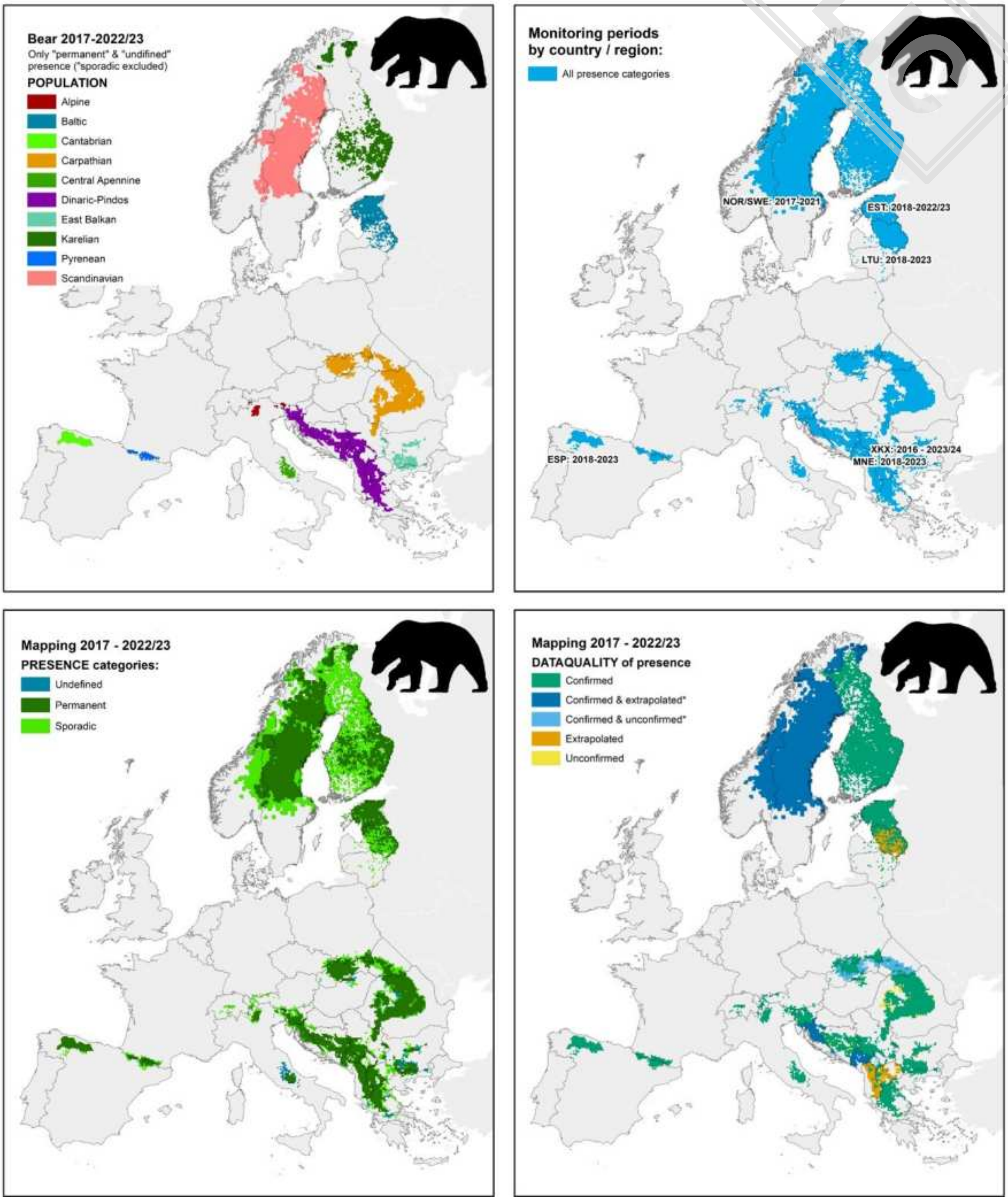

*Fig. 5: Brown bear populations, countries/regions with monitoring periods deviating from the 2017-2022/23 period, bear distribution, and underlaying data quality. *DATAQUAL: Mixed layer of confirmed and extrapolated or unconfirmed cells, where no separation at cell level was possible.*

Changes in brown bear distribution in Europe since 2012-2016

The brown bear distribution seems to have mainly gained. Looking at individual populations (Fig. 5 & 6, Table 3):

- The **Alpine population** lost some area, but mainly of the sporadic category. The population in Trentino has consolidated its permanent distribution but has a reduced sporadic distribution. There is less sporadic presence in the surrounding of Trentino and in Austria, but the sporadic distribution has spread into Germany. Italy self-reported an increasing distribution area of the bears in the Alps.
- The **Baltic population** has greatly gained in distribution, particularly in Latvia but also spreading into Lithuania. Both countries self-reported an increasing distribution area, whereas Estonia reported no obvious change.
- The distribution of the **Cantabrian population** has consolidated and expanded its the number of permanent cells, reaching a permanent connection between the Eastern and Western Cantabrian mountains and even sporadic presence towards Portugal. Spain self-reported an increasing distribution, and Portugal recorded its first bear visit in 2019.
- The **Carpathian population's** distribution has increased in all range countries. Gains in the north of the Carpathian arc are due to better monitoring data in Ukraine and first monitoring data from Hungary. A distinction between permanent and sporadic presence in Slovakia was possible due to better quality data. Range countries Romania and Poland self-reported no obvious change in the bear distribution, Ukraine a fluctuating trend, and Serbia, Slovakia and Hungary an increasing trend.
- The isolated **Central Apennine population's** distribution shows increased undefined presence primarily towards the north. Italy has self-reported an increasing distribution, but also a change in monitoring method.
- The **Dinaric-Pindos population's** distribution has increased and consolidated with improved permanent connectivity, although some gaps remain (e.g. between southern Croatia and Bosnia). Range countries Albania, Bosnia and Herzegovina, Croatia, Greece, and Serbia self-reported an increasing trend in the bear distribution, Bulgaria, Montenegro, and Slovenia no obvious change, and North Macedonia and Kosovo* an unknown trend in distribution.
- The **East Balkan population's** distribution has lost some area, and the current distribution suggests a new disconnection between the Stara Planina and Rodophi segments. Bulgaria self-reported a change in method and area losses may be due to changes in the way monitoring is conducted. Bulgaria self-reported a fluctuating range.
- The **Karelian population's** distribution seems to have lost some of the "permanent" bear area, especially along the Russian border, and some "sporadic" presence in the north. However, the country self-reported a change in method and an increasing distribution! Note that Finland does not buffer bear records in the way that Norway and Sweden do, creating an impression of a more limited permanent distribution when comparing to the neighbouring Scandinavian population.
- The **Pyrenean population** distribution area has increased and consolidated, and the connection between the west and eastern Pyrenees is now narrowly connected. The two range countries monitor the population's distribution jointly and self-reported an increasing range.
- The **Scandinavian population's** distribution has remained stable over the permanent distribution area and shown some losses and gains of sporadic presence along the fringes. Sweden and Norway jointly monitor the distribution and both self-reported no obvious change in the distribution.

Table 3: Changes in brown bear distribution in Europe since 2016, expressed as number of 10 x 10 km cells.

Population	N cells in 2016				N cells in 2022				Balance (%)			
	Permanent	Sporadic	Undefined	Total	Permanent	Sporadic	Undefined	Total	Permanent	Sporadic	Undefined	Total
Alpine	63	197		260	57	140		197	-10	-29	NA	**-24.2**
Baltic	407	60		467	596	315		911	46	425	NA	**95.1**
Cantabrian	88	53		141	160	27		187	82	-49	NA	**32.6**
Carpathian	947	137	183	1,267	1,286	327	43	1,656	NA	NA	NA	**30.7**
Central Apennine	80			80	44	1	56	101	NA	NA	NA	**26.3**
Dinaric-Pindos*	887	289	10	1,186	1,169	280	30	1,479	NA	NA	NA	**24.7**
East Balkan	218	186		404	180	97	70	347	-17	-48	NA	**-14.1**
Karelian	1,520	1,989		3,509	1,087	1,940		3,027	-28	-2	NA	**-13.7**
Pyrenean	36	29		65	78	59		137	117	103	NA	**110.8**
Scandinavian	2,489	1,584		4,073	2,539	1,395		3,934	2	-12	NA	**-3.4**
Total	**6,735**	**4,524**	**193**	**11,452**	**7,196**	**4,581**	**199**	**11,976**	**7**	**1**	**3**	**4.6**

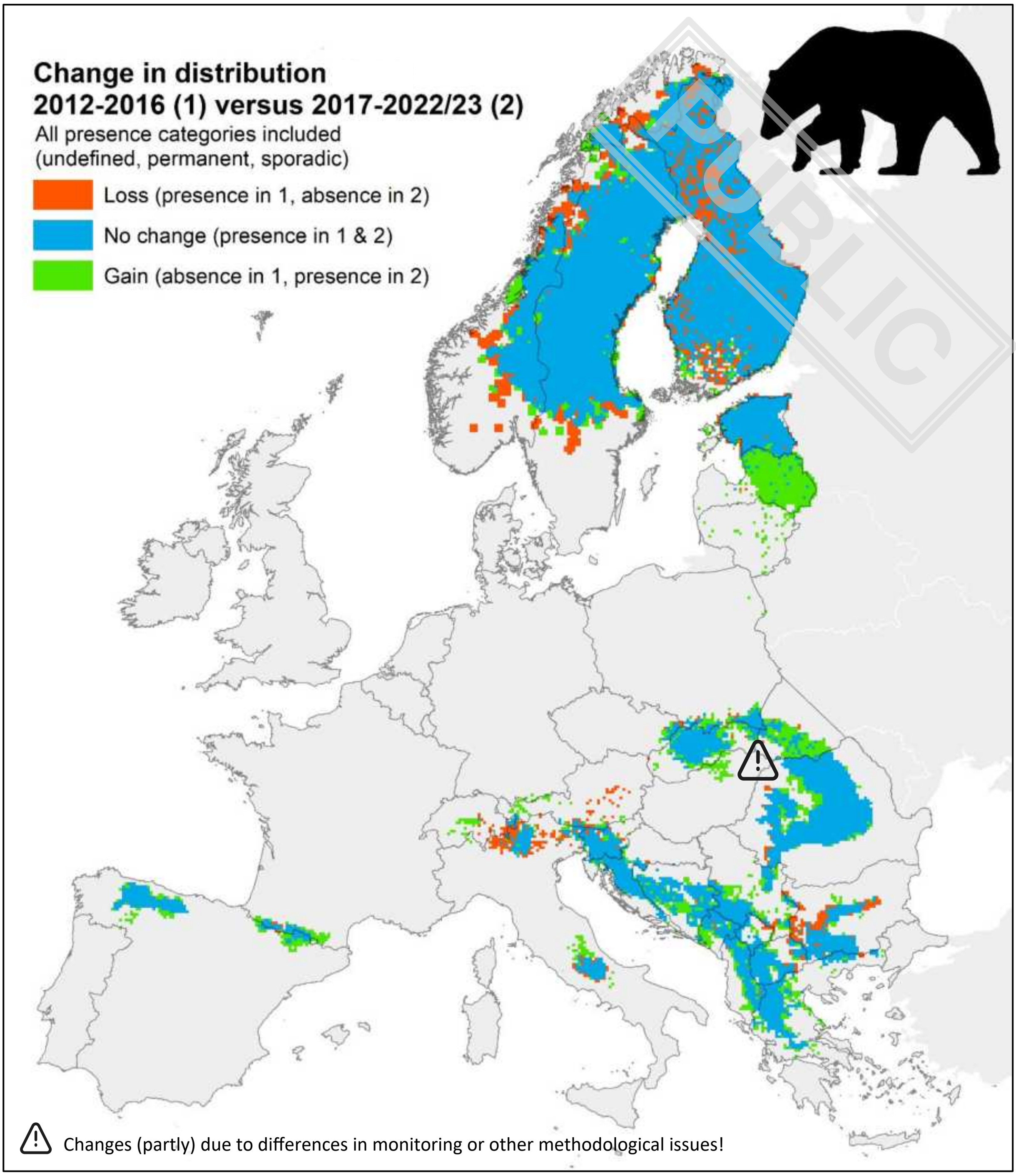

Fig. 6: Changes in brown bear distribution 2012-2016 versus 2017-2022/23.

3.1.2. Eurasian lynx

Overview of main distribution monitoring methods

The most important monitoring methods for determining lynx distribution are camera trapping, recording of SCALP C2 signs, questionnaires and interviews, family group observations, and active snow tracking. “Other” in Slovakia refers to opportunistic observations reported by hunters (Table 4).

Table 4: Main distribution monitoring methods for Eurasian lynx in Europe. BBA = Bohemian-Bavarian-Austrian population

Country	Range monitoring method												
	Dead animals	**Non-invasive genetics**	**Camera traps**	**GPS tracking**	**Active snow tracking**	**Howling surveys**	**Family groups**	**SCALP C2**	**Damage statistics**	**SCALP C3**	**Quest. & interviews**	**Past presence**	**Other***
Eurasian lynx													
Albania	<10%	<10%	>75%					<10%		10-25%	10-25%	<10%	
Austria - Alps	<10%		>75%								>75%		
Austria - BBA	<10%	<10%	>75%				25-50%				>75%		
Belgium	only single dispersers, by-catch of wolf monitoring												
Bosnia & Herzegovina		<10%	>75%	<10%	25-50%								
Bulgaria	only single dispersers, by-catch of wolf monitoring												
Croatia	>75%	<10%	>75%	10-25%	<10%		<10%	50-75%	>75%	<10%	>75%		
Czech Republic	<10%	<10%	>75%	<10%	25-50%			25-50%			>75%		
Estonia	<10%		10-25%		10-25%		25-50%	25-50%			>75%		
Finland	10-25%		10-25%		10-25%		>75%	>75%	10-25%		>75%		
France	>75%		>75%	<10%			>75%	>75%	>75%		>75%		
Germany			>75%	10-50%				>75%					
Germany - Alps	>75%							>75%		>75%	>75%		
Hungary	<10%	10-25%	50-75%		10-25%			>75%					
Italy		10-25%	>75%	10-25%			>75%	>75%					
Kosovo*	<10%		25-50%		10-25%			<10%	10-25%	50-75%	25-50%		
Latvia	>75%	<10%	<10%							10-25%			
Lithuania	<10%		25-50%					25-50%		<10%	>75%		
Montenegro	no information												
North Macedonia	<10%	<10%	50-75%	25-50%	<10%			<10%		<10%	<10%	<10%	
Poland	<10%	<10%	10-25%	10-25%	10-25%			10-25%	<10%		50-75%	<10%	
Romania			<10%		>75%			<10%		10-25%		>75%	
Serbia	<10%		25-75%	<10%							<10% & >75%		
Slovakia	>75%	<10%	>75%	10-25%	>75%		>75%	>75%	>75%	>75%	>75%		>75%
Slovenia	>75%	25-50%	>75%	50-75%	25-50%			>75%	>75%	>75%	50-75%	>75%	
Sweden & Norway	>75%		>75%		>75%		>75%		>75%				
Switzerland	>75%		>75%	<10%			>75%	>75%	25-50%	<10%			
Ukraine - Carpathians	<10%	<10%	<10%	<10%	10-25%	<10%	<10%	<10%	<10%	25-50%	<10%		
Ukraine - focal areas			>75%										

More details on species monitoring methods can be found in the compilation of the most recent literature on population and distribution estimates in Appendix 6.

Overview of main mapping methods

Distribution data was available for most countries up to 2022/23, with only Norway and Sweden presenting a one year shorter dataset (until 2021), and Kosovo* covering 10 years from 2014-2024. Most of the lynx presence is based on confirmed (C1 & C2) signs intersected directly with the 10 x 10 km grid.

Out of 30 countries/regions with lynx in our survey, 6 use buffers around lynx signs (Albania, Latvia, North Macedonia, Poland, Sweden, Norway) with 3 (Albania, North Macedonia, and Poland) additionally using modelling for extrapolated presence. Three countries (Croatia, Slovakia, Ukraine) have some, or most, information coming at the spatial scale of hunting grounds, which were intersected with the 10 x 10 km grid.

For 5 countries (Croatia, Sweden, Norway and in parts Slovakia and Ukraine) information on data quality was not available, or only partly available, at the individual cell level (Table 5).

Table 5: Mapping details for Eurasian lynx in Europe.

Country / Region	FINAL_time	Spatial scale	% Known range monitored: Active	% Known range monitored: Passiv	Large carnivore signs used	Definition of gridcells based on	Scale of data quality information	Presence categorisation based on	Method change	Range trend estimate since 2012-2016: Trend	Range trend estimate since 2012-2016: Assessment
Eurasian lynx											
Albania	2017 – 2022/23	Entire known range bu	100	20	C1 & C2	Buffered confirmed presence signs & modelling overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real & method change
Austria - Alps	2017 – 2022/23	Entire known range	20	50	C1	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Fluctuating	Real
Austria - BBA	2017 – 2022/23	Entire known range	75	ınknowr	C1	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Yes	Fluctuating	Real
Belgium	2020 – 2022/23	Only single dispersers present			C1	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	No distinction was made	NA	First dispersers	Real
Bosnia & Herzegovina	2017 – 2022/23	Entire known range	75	15	C 1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Fluctuating	Real
Bulgaria	2017 – 2022/23	Only single dispersers present			C1 & C3	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	No distinction was made	No	Only single individuals	Real
Croatia	2018 – 2023	Entire known range	80	20	C1 & C2	Hunting grounds with confirmed presence signs overlaid with the 10 x 10 grid	Country level	Re-occurring presence and/or reproduction	Yes	Increasing	Real
Czech Republic	2017 – 2022/23	Entire known range	80	20	C 1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	No obvious change	Real
Estonia	2018 – 2022/23	Entire known range	100	100	C 1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	No obvious change	Real
Finland	2017 – 2022/23	Entire known range	100	80	C 1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Reproduction only	Yes	Fluctuating	Real
France	2017 – 2022/23	Entire known range	?	100	C 1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Yes	Increasing	Real
Germany	2017 – 2022/23	Entire known range	20 - 100 (depending on federal state)		C 1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Germany - Alps	2017 – 2022/23	Entire known range	0	100	C 1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	No obvious change	Real
Hungary	2017 – 2022/23	Entire known range	80	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	No obvious change	Real
Italy	2017 – 2022/23	Entire known range	75	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Yes	Increasing	Real
Kosovo*	2014 – 2024	Entire known range	60	40	C1, C2&C3	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Fluctuating	More data needed
Latvia	2017 – 2023	Entire known range	<5	100	C1	Buffered confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Mortality	Yes	Increasing	Method change
Lithuania	2018 – 2023	Entire known range	100	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real & method change
Montenegro	2018 – 2023	No information	60	60	C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	No information			
North Macedonia	2017 – 2022/23	Only reference areas	50	30	C1 & C2	Buffered confirmed presence signs & modelling overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	No obvious change	Real
Poland	2017 – 2022/23	Entire known range	20	80	C1 & C2	Buffered confirmed presence signs & modelling overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Yes	Increasing	Real
Romania	2017 – 2022/23	Entire known range	70	30	C1	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	No obvious change	Real
Serbia	2017 – 2022/23	Entire known range	30	70	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	No obvious change	Real
Slovakia	2017 – 2022/23	Entire known range	100	100	C1, C1*-C3	Hunting grounds with confirmed presence signs overlaid with the 10 x 10 grid & additional C1 & C2	Cell & country level	Re-occurring presence and/or reproduction	No	No obvious change	Real
Slovenia	2019 – 2022/23	Entire known range	95	5	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Yes	Increasing	Real
Switzerland	2017 – 2022/23	Entire known range	100	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Yes	Increasing	Real
Ukraine - Carpathians	2017 – 2023 (focus 2019)	Carpathians	10	100	C1*-C3	Hunting grounds with confirmed presence signs overlaid with the 10 x 10 grid & additional C1 & C2	Cell & country level	Hunter density estimate	Yes	Fluctuating	Real
Ukraine - focal areas	2020-2023	Only reference areas	NA	25 - 50	C1	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Data used for data quality only	NA	Unknown	More data needed
Sweden	2017 – 2021	Entire known range	100	100	C1* & C2	Buffered confirmed presence signs overlaid with the 10 x 10 grid	Country level	Reproduction only	No	Increasing	Real
Norway	2017 – 2021	Entire known range	100	100	C1* & C2	Buffered confirmed presence signs overlaid with the 10 x 10 grid	Country level	Reproduction only	No	No obvious change	Real

The distinction between permanent and sporadic presence was primarily made based on re-occurrence and/or reproduction. However, 3 countries (Finland, Sweden, Norway) used exclusively reproduction to define the permanent distribution, 2 with only a few dispersers made no distinction

(Belgium, Bulgaria), 1 used mortality events (Latvia), 1 used hunter derived density estimates (Ukraine), and 1 did not provide any information (Montenegro).

The trend in Eurasian lynx distribution was estimated to be increasing in 11 countries/regions, showing no obvious change in 9, and fluctuating in 6. Two countries had only single dispersers and in one the trend was unknown. No country/region reported a decreasing trend in distribution area (Table 5).

Current Eurasian lynx distribution in Europe

The lynx is currently found in 25 of the 34 countries/regions surveyed. The species is absent or only found in border cells in Denmark, Greece, Kosovo*, Luxembourg, Montenegro, Portugal, Spain, The Netherlands, and Turkey. The possible presence in Greece in the 2012-2016 lynx map could not be confirmed and was excluded from the distribution area gain/loss calculation table (Fig. 7, Table 6).

The total distribution area encompassed by the Eurasian lynx in Europe currently covers ca. 1.47 million km^2, which is a 21.2% increase in distribution since 2016 (Table 6).

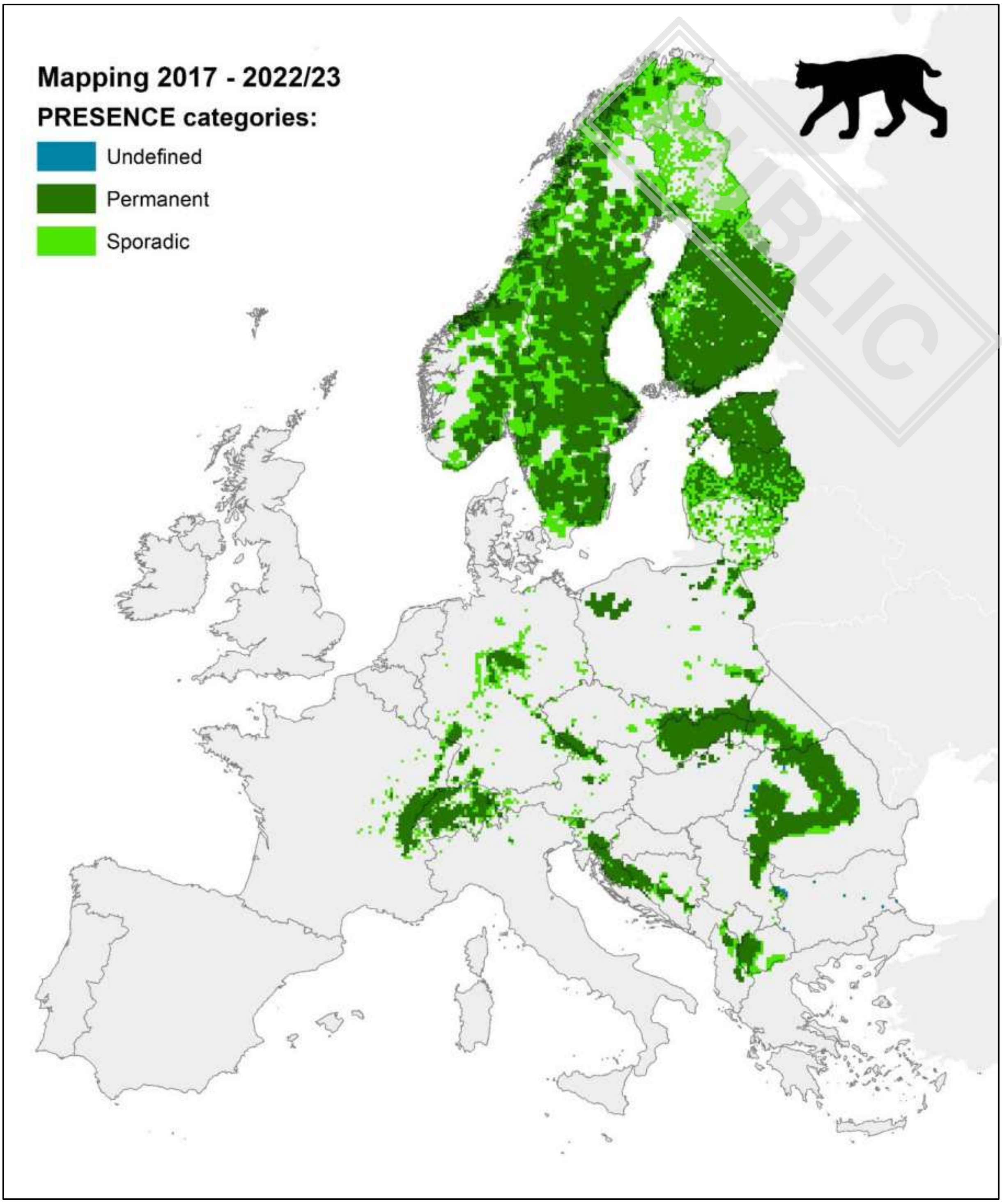

Fig. 7: Eurasian lynx distribution in Europe for the period 2017-2022/23.

Most of the lynx distribution area in all populations is based on confirmed lynx signs, often in combination with some form of extrapolation (buffers and/or modelling, see Table 5). The small Balkan population still has the least robust data foundation. In parts of the Carpathian population monitoring is dependent on observations from hunting grounds or protected areas (in Ukraine and Slovakia). These observations are often less formally documented or accessible and include direct observations and thus were given the mixed status of "Confirmed and unconfirmed" data quality. Where available, this data was confirmed with C1 & C2 data from dedicated monitoring projects, especially those using camera trapping or non-invasive genetic monitoring. In Latvia buffered C1 records, which were exclusively used in the past, are now supplemented with C3 records from observations recorded by volunteers via wildlife citizen science apps, so that the distribution is now filled in with sporadic presence based on unconfirmed records (Fig. 8).

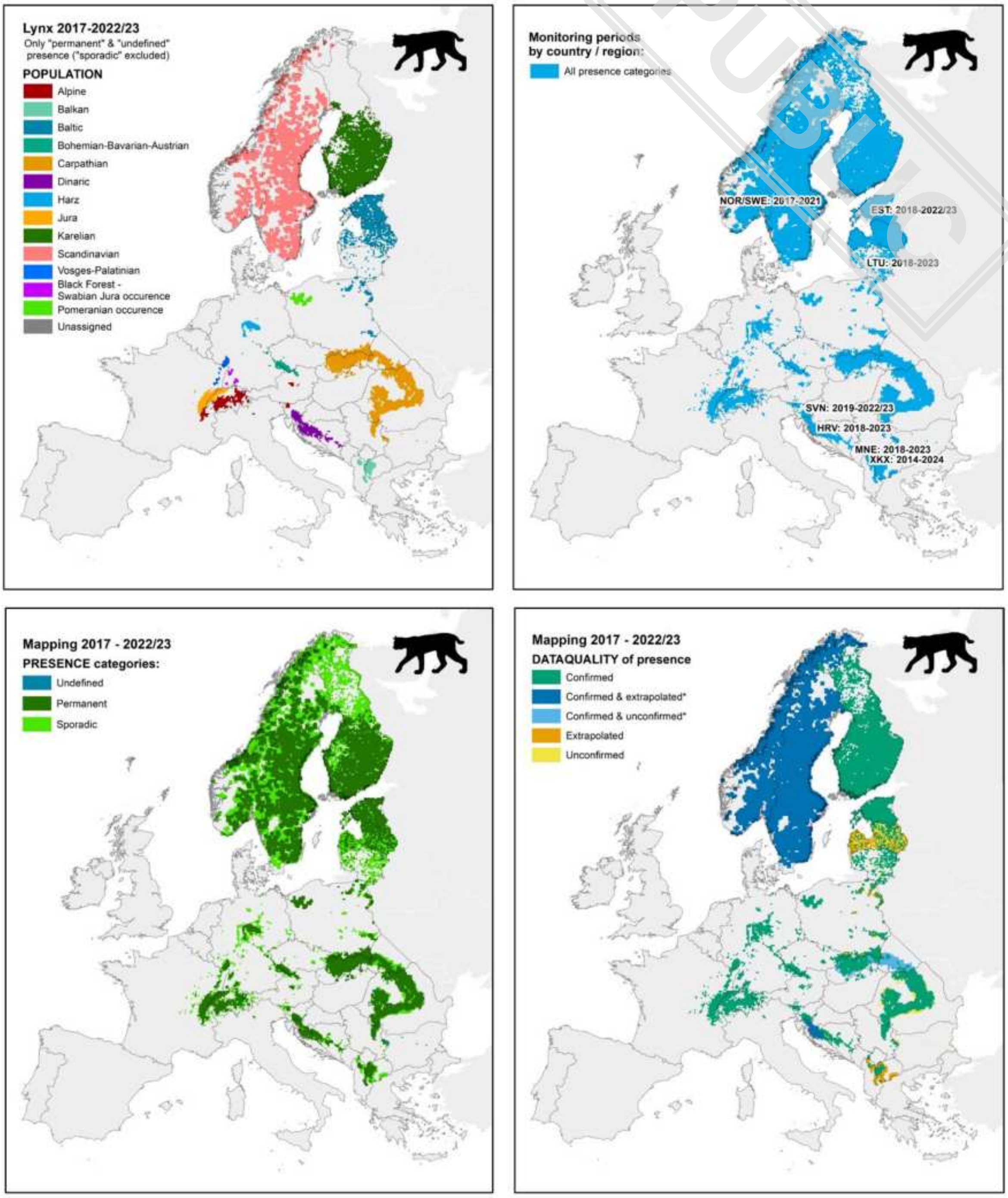

*Fig. 8: Eurasian lynx populations, countries/regions with monitoring periods deviating from the 2017-2022/23 period, lynx distribution, and underlaying data quality. *DATAQUAL: Mixed layer of confirmed and extrapolated or unconfirmed cells, where no separation at cell level was possible.*

Change in lynx distribution 2012-2016 versus 2017-2022/23

The Eurasian lynx distribution seems to have mainly gained area due to natural expansion, active reinforcement (translocation) of reintroduced populations, and new reintroduction projects. For countries having several population segments, self-reported trend estimates were not necessarily available for each segment, but rather for the overall national population. Looking at individual populations (Fig. 8 & 9, Table 6):

- The **Alpine population's** distribution has expanded in both the eastern and western Alps and is now connected via permanent cells to the Jura population and via sporadic cells to the new Black Forest – Swabian Alp occurrence. In the eastern Alps the area of permanent distribution of the Alpine lynx population remains still largely separated from the Dinaric population, although sporadic cells are documented. The distribution map for the Alpine population is regularly updated within the SCALP initiative (Molinari-Jobin et al. 2021). The range countries largely self-reported increasing trends for the distribution, but a fluctuating trend was reported for the Austrian Alps and no obvious change for the German Alps.
- The isolated **Balkan population's** distribution appears to have increased, but large parts of this expansion are based on model extrapolations. Permanent presence has been primarily documented in North Macedonia and a small part in Albania. The loss in Greece is due to the failure to confirm possible presence mapped in 2012-2016 (the distribution change is shown in the change map but was not included in the area calculation table). Albania self-reports an increasing distribution trend which is due to both a change in method and a real change, but North Macedonia reported no obvious change.
- The **Baltic population's** distribution has increased. The large increase in Latvia is due to a change in method for recording sporadic presence and a real change for permanent presence. The distribution is also increasing in Lithuania but has not changed much in eastern Poland, apart from an increase in sporadic occurrence cells in the isolated Roztocze region.
- The **Bohemian-Bavarian-Austrian population's** distribution has expanded to both the northwest and to the southwest and seems about to connect with the Harz population range. However, the distribution area is self-reported as fluctuating in Austria and showing no real change in the Czech Republic. For Germany no assessment was available for this population segment separately.
- The **Carpathian population's** distribution seems stable. Gains in the north of the Carpathian arc are due to better monitoring data in Ukraine and the inclusion of the first monitoring data from Hungary. The range countries largely self-report no obvious change in the distribution (Romania, Slovakia, Serbia) or a fluctuating range (Ukraine). For Poland no assessment was available for this population segment separately.
- The **Dinaric population's** distribution has changed very little and remains separated from the Alpine population and Balkan population. Slovenia and Croatia self-reported an increasing distribution (there has been extensive population reinforcement in recent years (Fležar et al. 2024)), and Bosnia and Herzegovina a fluctuating distribution.
- The distribution of the **Harz population** has increased and is starting to connect with the Bohemian-Austrian-German and the Vosges-Palatinian populations. The overall distribution of lynx in Germany was self-reported as increasing.
- The **Jura population's** distribution has increased and is now connected via permanent cells to the Alpine population. Connections to the Vosges-Palatinian and the Black Forest – Swabian Jura also seem to be developing. In addition, there is more sporadic presence westwards. The distribution of the lynx in Switzerland and France was self-reported as increasing.

- The **Karelian population's** distribution has remained stable, but shows more sporadic presence in the north, which is likely due to a change in monitoring methods. The distribution is self-reported as fluctuating by Finland.
- The **Scandinavian population's** distribution has remained stable, but the area of permanent distribution has consolidated in Sweden. Sweden self-reported an increase in the distribution while Norway reported no obvious change.
- The **Vosges-Palatinian population's** distribution has increased due to a reintroduction program in the Palatine Forest in Germany (Port et al. 2024) and likely due to expansion of the Jura population distribution.
- A new reintroduction project reintroduced 61 captive born lynx in NW Poland between 2019-2021 (Skorupski et al. 2022). Multiple lynx in this new **Pomeranian occurrence** have been followed by GPS tracking and have roamed over quite a large area and long-distance dispersal has occurred towards the southwest into a new area in Brandenburg and Saxony in Germany.
- Dispersing lynx have reached the Black Forest and the Swabian Jura in the past. This has resulted in an increase and permanent presence in the **Black Forest – Swabian Jura** occurrence area; however no female lynx have been documented so far. Steps to strengthen this occurrence have started in December 2023 with the first release of a female lynx.

Table 6: Changes in the distribution of Eurasian lynx in Europe since 2016, expressed as number of 10 x 10 km cells.

Population	2016				2022				Balance (%)			
	Permanent	Sporadic	Undefined	Total	Permanent	Sporadic	Undefined	Total	Permanent	Sporadic	Undefined	Total
Alpine	147	151		298	239	143		382	62.6	-5.3		28.2
Balkan*	48	111	2	161	105	90		195	118.8	-18.9	NA	21.1
Baltic	716	252		968	992	674		1,666	38.5	167.5		72.1
Bohemian-Bavarian-Austrian	59	53		112	89	45		134	50.8	-15.1		19.6
Carpathian	1,204	158		1,362	1,415	249	24	1,688	17.5	57.6		23.9
Dinaric	217	43		260	210	59		269	-3.2	37.2		3.5
Harz	39	90		129	79	158		237	102.6	75.6		83.7
Jura	139	34		173	169	53		222	21.6	55.9		28.3
Karelian	2,269	323		2,592	1,887	1,053		2,940	-16.8	226.0		13.4
Scandinavian	4,523	1,295		5,818	4,043	2,492		6,535	-10.6	92.4		12.3
Vosges-Palatinian	5	72		77	49	51		100	880.0	-29.2		29.9
Pomeranian occurrence		3		3	91			91	NA	NA	NA	NA
Black Forest-Swabian Jura		37		37	30	26		56		-29.7		51.4
Unassigned	3	19		32	5	45	2	52	66.7	136.8		62.5
Total	**9,369**	**2,641**	**2**	**12,022**	**9,403**	**5,138**	**26**	**14,567**	0.4	94.5	NA	**21.2**

*The cells defined as "possible presence" in the maps for 2016-2012 for the portion of the Balkan lynx distribution in Greece could not be confirmed and we now assume an absence of lynx in Greece for both mapping periods and these 111 cells of potential presence were excluded from the balance calculation in this table.

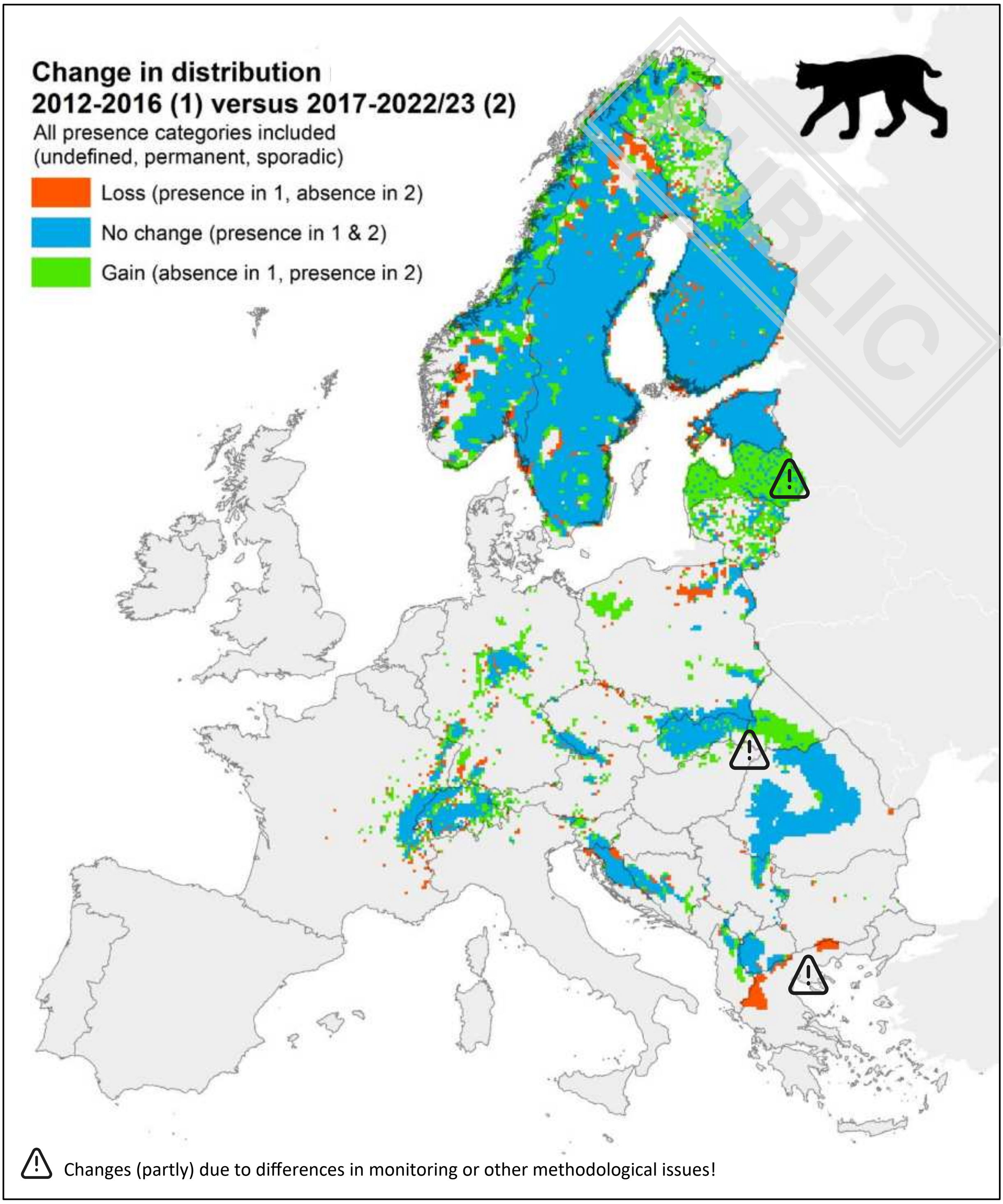

Fig. 9: Changes in Eurasian lynx distribution 2012-2016 versus 2017-2022/23.

3.1.3. Wolf

Overview of main distribution monitoring methods

The most important monitoring methods for determining wolf distribution are dead animals, non-invasive genetic monitoring (scats, urine), camera trapping, active snow tracking, and recording of other SCALP C2 signs. "Other" in Slovakia refers to opportunistic observations reported by hunters (Table 7).

Table 7: Main distribution *monitoring method for the wolf.*

Country	Range monitoring method												
	Dead animals	Non-invasive genetics	Camera traps	GPS tracking	Active snow tracking	Howling surveys	Family groups	SCALP C2	Damage statistics	SCALP C3	Quest. & interviews	Past presence	Other*
Wolf													
Albania	<10%		25-50%			<10%				<10%	<10%	<10%	
Austria	<10%	>75%	<10%	<10%			<10%				>75%		
Belgium	<10%	>75%	>75%					10-25%	50-75%				
Bosnia & Herzegovina	50-75%		25-50%		<10%	<10%		50-75%			<10%		
Bulgaria	>75%		50-75%	<10%	<10%	<10%	<10%	50-75%		<10%			
Croatia	>75%	50-75%	25-50%	10-25%	<10%	<10%	<10%	>75%	>75%	<10%			
Czech Republic	<10%	50-75%	50-75%	10-25%	50-75%	<10%		25-50%	<10%		>75%		<10%
Denmark	<10%	25-50%	25-50%	<10%	<10%		10-25%	<10%	<10%		>75%		
Estonia	<10%		10-25%	<10%		<10%	10-25%	10-25%			>75%		
Finland	10-25%	>75%	10-25%	10-25%	10-25%		>75%	>75%					
France	<10%	25-50%	25-50%		10-25%	<10%		25-50%					
Germany	>75%	>75%	>75%	<10%	10-25%			>75%			>75%		
Greece	<10%		10-25%	<10%		10-25%	<10%	<10%	>75%		<10%		
Hungary	10-25%	10-25%	50-75%		10-25%	<10%	10-25%	>75%	<10%				
Italy - Alps	>75%	>75%	>75%		>75%			>75%					
Italy - Peninsula	>75%	>75%	>75%		>75%			>75%					
Kosovo*	no dedicated monitoring - by-catch from lynx camera trap monitoring												
Latvia			<10%		10-25%				10-25%	10-25%			
Lithuania	25-50%		10-25%					10-25%		<10%	>75%		
Luxembourg	only single dispersers, detected through analysis of livestock kills and photos/videos from the public												
Montenegro	no information												
North Macedonia	<10%	<10%	25-50%		<10%	<10%	<10%	<10%	<10%	<10%	10-25%	<10%	
Poland	10-25%	10-25%	10-25%	<10%	10-25%	<10%	10-25%	10-25%	25-50%		50-75%	<10%	
Portugal	<10%	>75%	>75%			>75%	>75%	>75%	>75%			>75%	
Romania			<10%		>75%			<10%	<10%	<10%		>75%	
Serbia	25-50%		25-50%	<10%					<10%				
Slovakia	>75%	10-25%	>75%	25-50%	>75%	<10%	>75%	>75%	>75%	>75%	>75%		>75%
Slovenia	<10%	50-75%							10-25%		>75%		10-25%
Spain	>75%	<10%	>75%	10-25%	<10%	>75%			>75%				
Sweden & Norway	>75%	>75%	>75%		>75%				>75%				
Switzerland	>75%	>75%	50-75%	<10%		25-50%	>75%	>75%	>75%				
The Netherlands	<10%	>75%	25-50%		<10%	<10%		25-50%	>75%	<10%	<10% & >75%		
Turkey - European part												>75%	<10%
Ukraine - Carpathians	<10%	<10%	<10%	<10%	10-25%	<10%	<10%	<10%	<10%	25-50%	<10%		

Overview of main mapping methods

Distribution data were available for most countries up to 2022/23. However, France reported only until 2019/20, Italy for the peninsula area reported for one intensive monitoring period in 2020-2021 (Gervasi et al. 2024), Norway and Sweden reporting from 2017-2021, and Turkey presented new data based on media reports with video and/or photo documentation only for the last 2 years, but was otherwise based on older data (Ambarli et al. 2016).

Out of 35 countries/regions with wolf presence, 7 use buffers around wolf signs (Albania, Bulgaria, Latvia, North Macedonia, Poland, Sweden, Norway) with 3 (Albania, Italy – Peninsula, North Macedonia, and Poland) additionally using modelling for extrapolated presence. Three countries (Croatia, Slovakia, Ukraine) used some or most wolf signs obtained at the level of hunting grounds, which were intersected with the 10 x 10 km grid.

For 5 countries (Croatia, Sweden, Norway and in part Slovakia and Ukraine) information on data quality was not, or only partly, available at the individual cell level.

A distinction between permanent and sporadic presence was primarily made based on re-occurrence and/or reproduction. However, 2 countries (Sweden, Norway) exclusively used reproduction, and 1 country (Latvia) exclusively used mortality (locations of dead animal) to define the area of permanent distribution. Three countries/regions made no distinction between permanent and sporadic (Greece, Italy – Peninsula, and Turkey). Kosovo* used data quality, Romania a combination of data quality and quantity, Montenegro used geographic location, Ukraine used hunter density estimates <1/100km^2, and North Macedonia did not specify.

The distribution area of wolves was estimated to have increased in 20 countries/regions, showed no obvious change in 6, fluctuated in 2, decreased in 2 (Bosnia and Herzegovina, Portugal), and consisted of only single individuals in 1, and was unknown or unreported from 5 (Table 8).

Table 8: Mapping details for the wolf in Europe.

Country / Region	FINAL_time	Spatial scale	% Known range monitored		Large carnivore signs used	Definition of gridcells based on	Scale of data quality information	Presence categorisation based on	Method change	Range trend estimate since 2012-2016	
			Active	Passiv						Trend	Assessment
Wolf											
Albania	2017 – 2022/23	Only reference areas	15	20	C1	Buffered confirmed presence signs & modelling overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Yes	No obvious change	Real
Austria	2017 – 2022/23	Entire known range	10	90	C1 (C2)	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Belgium	2017 – 2022/23	Entire known range	100	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Bosnia & Herzegovina	2017 – 2022/23	Entire known range bu	85	10	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Decreasing	Real & method change
Bulgaria	2017 – 2022/23	Entire known range	variable	100	C1*-C3	Buffered confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Yes	Fluctuating	Real
Croatia	2017 – 2022/23	Entire known range	80	20	C1 & C2	Hunting grounds with confirmed presence signs overlaid with the 10 x 10 grid	Country level	Re-occurring presence and/or reproduction	No	Increasing	Real
Czech Republic	2017 – 2022/23	Only reference areas	75	5	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Yes	Increasing	Real
Denmark	2017 – 2022/23	Entire known range	100	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Estonia	2018 – 2022/23	Entire known range	100	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	No obvious change	Real
Finland	2017 – 2023/24	Entire known range	100	80	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Yes	Increasing	Real
France	2017 – 2019/20	Entire known range	42	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Germany	2017 – 2022/23	Entire known range	20 - 100 (depending on federal state)		C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Greece	2019 – 2023	Entire range with C2/C	25	25	C1 & C2, some C3	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	No distinction was made	No	Increasing	Real & method change
Hungary	2017 – 2022/23	Entire known range	75	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Italy - Alps	2020 - 2022/23	Entire known range	100	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Italy - Peninsula	2020 – 2021	Entire known range	100 (randomly sampled)		C1 & C2	Confirmed presence signs and modelling overlaid with the 10 x 10 grid	Cell level	No distinction was made*	Yes	Increasing	Real & method change
Kosovo*	2016 - 2023/24	no information			C1 & C2-C3	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Data quality	no information		
Latvia	2017 – 2023	Entire known range	100	0	C1	Buffered confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Mortality	yes	Increasing	Real & method change
Lithuania	2018 – 2023	Entire known range	100	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence	No	Increasing	Real & method change
Luxembourg	2017 – 2023	Only single dispersers present			C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Only first individuals	
Montenegro	2022 – 2023	No information	60	60	C1	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Geographic location	no information		
North Macedonia	2017 – 2022/23	Monitoring is opportu	0	25	C1 & C2	Buffered confirmed presence signs & modelling overlaid with the 10 x 10 grid	Cell level	Not specified	No	Unknown	More data needed
Poland	2017 – 2022/23	Entire known range	20	80	C1 & C3	Buffered confirmed presence signs & modelling overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Portugal	2017 – 2022/23	Entire known range	100	0	C1 & C2	Confirmed presence signs or UTM grid overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Yes	Decreasing	Real
Romania	2017 – 2022/23	Entire known range	70	30	C1, C2 & C3	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Data quality & Number of signs	Yes	No obvious change	Method change
Serbia	2017 – 2022/23	Entire known range	60	40	C1	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	No obvious change	Real
Slovakia	2017 – 2022/23	Entire known range	100	100	C1, C1*-C3	Hunting grounds with confirmed presence signs overlaid with the 10 x 10 grid & additional C1 & C2	Cell & country level	Re-occurring presence and/or reproduction	No	Increasing	Real
Slovenia	2017 – 2022/23	Entire known range	70	30	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Yes	Increasing	Real
Spain	2017 – 2022/23	Entire known range	90	10	C1* & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	No obvious change	Expansion of non-breeders
Switzerland	2017 – 2022/23	Entire known range	100	100	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	Slight	Increasing	Real
The Netherlands	2017 – 2022/23	Entire known range	80	20	C1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	Re-occurring presence and/or reproduction	No	Increasing	Real
Turkey - European part	<2016 & 2023/2024	European part	unkown	ınknowr	(C1), C3	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	No distinction was made	NA	Unknown	
Ukraine - Carpathians	2017 – 2023 (2019)	Carpathians	10	100	C1*-C3	Hunting grounds with confirmed presence signs overlaid with the 10 x 10 grid & additional C1 & C2	Cell & country level	Hunter density estimate (>1/km2) for permanent	Yes	Fluctuating	Real
Sweden	2017 – 2021	Entire known range	100	100	C1 & C2	Buffered confirmed presence signs overlaid with the 10 x 10 grid	Country level	Reproduction only	No	Increasing	Real
Norway	2017 – 2021	Entire known range	100	100	C1 & C2	Buffered confirmed presence signs overlaid with the 10 x 10 grid	Country level	Reproduction only	No	No obvious change	Real

Current wolf distribution in Europe 2017-2022/23

The wolf is currently found in all 34 of the 34 countries/regions monitored (Fig. 10). The total distribution encompassed by the wolf in Europe currently covers ca. 2.2 million km^2, which is a 40% increase since 2016 (Fig. 10, Table 9).

Although there is no doubt that wolf distribution has greatly increased, some of this increase can be clearly attributed to a change in methods, particularly for the Italian peninsula (gain: 63,200 km^2), Latvia (gain: 49,000 km^2), and access to new data from Ukraine (gain: 27,500 km^2). The gain by these 3 countries alone accounts for 22% of the range change (Table 9).

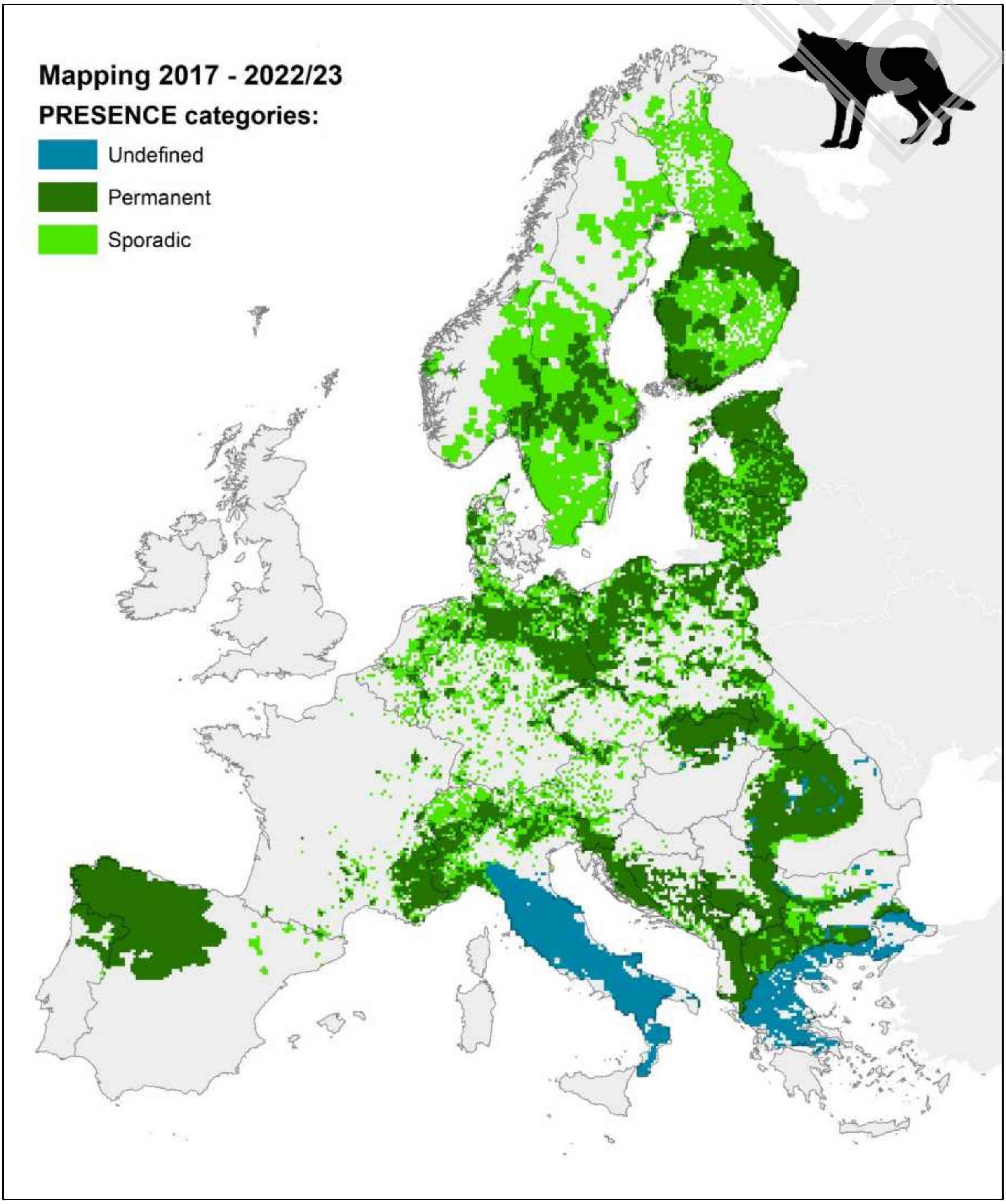

Fig. 10: Wolf distribution in Europe for the period 2017-2022/23.

Most of the wolf distribution area in all populations is based on confirmed wolf signs, often in combination with some form of extrapolation (buffers and/or modelling, see Table 2). Only in some parts of the Dinaric-Balkan region is the wolf distribution largely based on extrapolated or unconfirmed records (Albania, North Macedonia, and Turkey). In parts of the Carpathian population monitoring is dependent on observations from hunting grounds or protected areas in Ukraine and Slovakia. These observations are often less formally documented or less accessible, and include direct observations, and were given the mixed status of "Confirmed and unconfirmed" data quality. Where available, this data was confirmed with C1 & C2 data from dedicated monitoring projects, especially those using camera trapping or non-invasive genetic monitoring (Fig. 11).

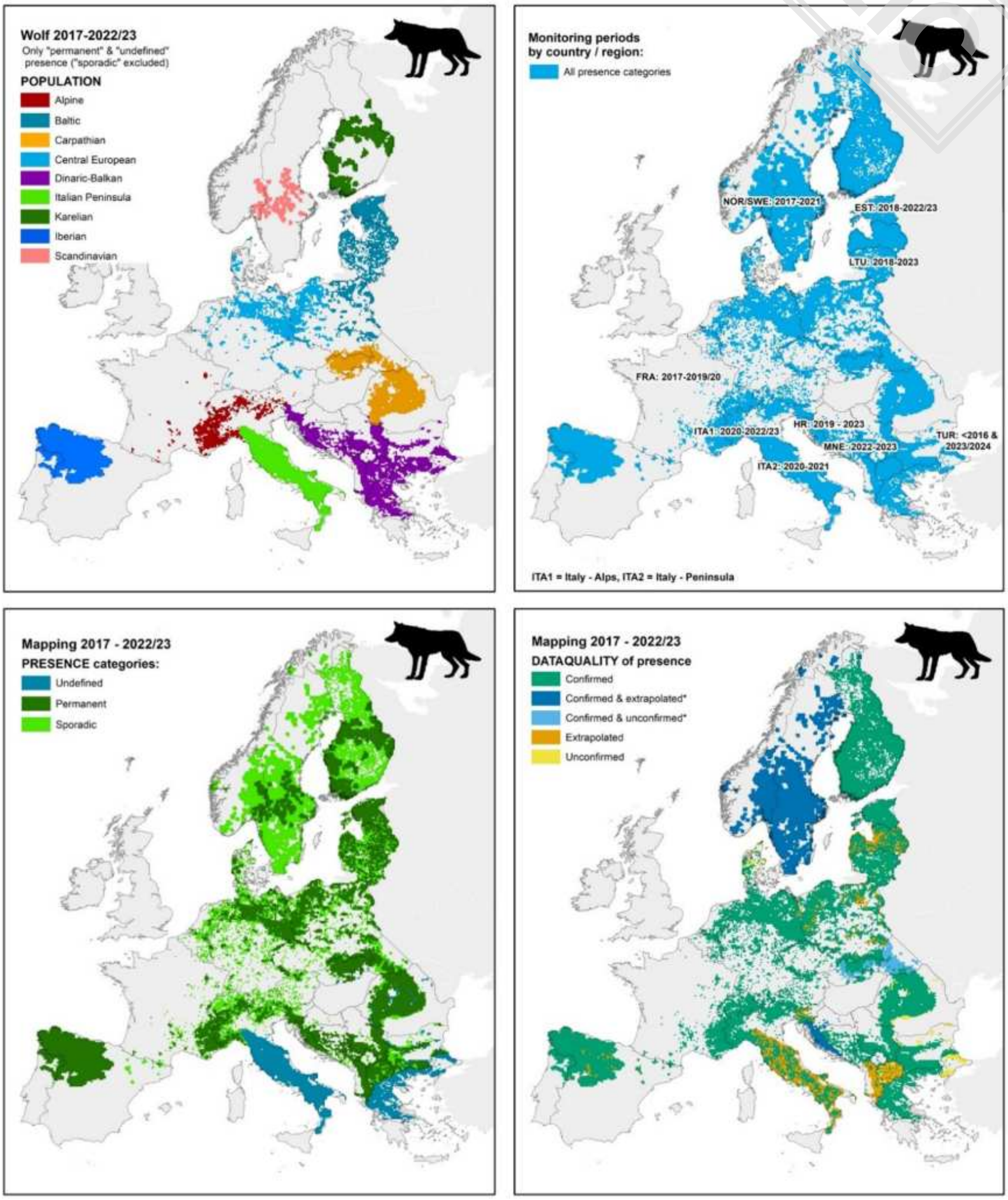

*Fig. 11: Wolf populations, countries/regions with monitoring periods deviating from the 2017-2022/23 period, wolf distribution, and underlaying data quality. *DATAQUAL: Mixed layer of confirmed and extrapolated or unconfirmed cells, where no separation at cell level was possible.*

Change in wolf distribution 2012-2016 versus 2017-2022/23

The distribution area of wolves increased due to natural expansion. For countries having several population segments, self-reported trend estimates were not necessarily available for each segment but rather the national population. Looking at individual populations (Fig. 11 & 12, Table 9):

- The **Alpine population's** distribution has increased, and the area of permanent presence is consolidating especially in Italy, France, and Switzerland. The population's distribution is in the process of connecting with the Central European to the north and the Dinaric-Balkan population to the east, and wolves of Alpine origin have maintained a presence in the Pyrenees and soon may be able to connect to the Iberian wolf population. Monitoring is increasingly happening at the population level (Marucco et al. 2023). The main range countries all self-reported an increasing population range.
- The **Baltic population's** distribution is increasing, although the apparent increase in the sporadic presence in Latvia is largely due to methodical changes (the 2012-2016 mapping did not include buffers and unconfirmed data). Latvia and Lithuania self-report an increase in the wolf distribution, Estonia reported no obvious change, and Poland reported an increasing population distribution at the overall national level.
- The **Carpathian population's** distribution has remained stable, except for the north where the area of permanent presence has increased widely in Slovakia (but this may also have to do with access to better data for the current mapping), increased somewhat in Poland, and is also reaching into Hungary. The apparent gains in Hungary and Ukraine are due to better monitoring data in Ukraine and the first inclusion of monitoring data from Hungary. The 3 northern countries all self-report an increasing trend in wolf distribution area. Romania self-reported no obvious change.
- The **Central European population's** permanent distribution has greatly expanded and consolidated in northwest Poland and northeast Germany and now stretches into the Netherlands, Belgium, and Denmark, and along the border with the Czech Republic. Monitoring methods are increasingly harmonised between the range countries (Reinhardt et al. 2015) as is online reporting for Germany and the Benelux countries. Sporadic occurrence is starting to connect to the Alpine, Dinaric-Balkan, and Carpathian populations. All range countries self-reported an increasing population range.
- The **Dinaric-Balkan population's** distribution has remained largely stable but shows some more losses than gains. Better connectivity is suggested in Montenegro, but this is an artefact as no data was available from this country for 2012-2016. The largest distribution loss is visible in northern Croatia and northeast Bulgaria. For Bulgaria this may be due to a change in methods and monitoring effort. Bulgaria self-reported a fluctuating trend, Albania and Serbia reported no obvious trend, Bosnia-Herzegovina a decreasing trend and Kosovo*, Montenegro, and North Macedonia have no information and need more data. Only Croatia and Slovenia self-reported an increasing trend in the wolf distribution area.
- The **Italian Peninsula population's** distribution was mapped for the first time in its entirety in a standardised and representative way combining field inspections and modelling (Gervasi et al. 2024).Consequently, the maps from 2012-2016 and now are not directly comparable. The wolf distribution area on the Italian peninsula is now fully connected with the Alpine wolf population. The distribution in the Italian Peninsula is self-reported to have increased due to the combination of a change in method and a real trend.
- The **Karelian population's** distribution has increased both in permanent and sporadic cells. Finland self-reported an increasing area of distribution.

- The **Iberian population's** distribution has remained stable in Spain but lost some area in Portugal. In line with this, Portugal self-reported a decreasing distribution and Spain no obvious change, but an expansion of non-breeders.
- The **Scandinavian population's** permanent distribution has decreased but may have also been overestimated for the period 2012-2017 (G. Chapron pers. comm. 2023). An enforced zoning policy does not allow for expansion of the permanent wolf distribution in Norway and not surprisingly Norway self-reported no obvious change. Sweden, on the other hand, self-reported an increasing wolf distribution which is likely due to the expansion of the area of sporadic presence, especially in the south.

Table 9: Changes in the wolf distribution in Europe since 2016, expressed as number of 10 x 10 km cells.

Population	N cells in 2016				N cells in 2022				Balance (%)			
	Permanent	Sporadic	Undefined	Total	Permanent	Sporadic	Undefined	Total	Permanent	Sporadic	Undefined	Total
Alpine	536	517		1,053	853	896		1,749	59.1	73.3		66.1
Baltic	1,271	324		1,595	1,770	770		2,540	39.3	137.7		59.2
Carpathian	1,229	122	202	1,553	1,670	286	70	2,026	35.9	134.4	-65.3	30.5
Central European	487	520		1,007	1,669	1,703		3,372	242.7	227.5		234.9
Dinaric-Balkan	2,277	702	18	2,997	1,678	353	817	2,848	-26.3	-49.7	4438.9	-5.0
Iberian	1,199	140	169	1,508	1,578	61		1,639	31.6	-56.4	-100.0	8.7
Italian Peninsula	531	203		734	6	12	1,348	1,366	-98.9	-94.1		86.1
Karelian	510	1,378		1,888	1,176	1,717		2,893	130.6	24.6		53.2
Scandinavian	1,518	2,030		3,548	635	3,136		3,771	-58.2	54.5		6.3
Unassigned		7		7					NA	NA		NA
Total	**9,558**	**5,943**	**389**	**15,890**	**11,035**	**8,934**	**2,235**	**22,204**	**15.5**	**50.3**	**474.6**	**39.7**

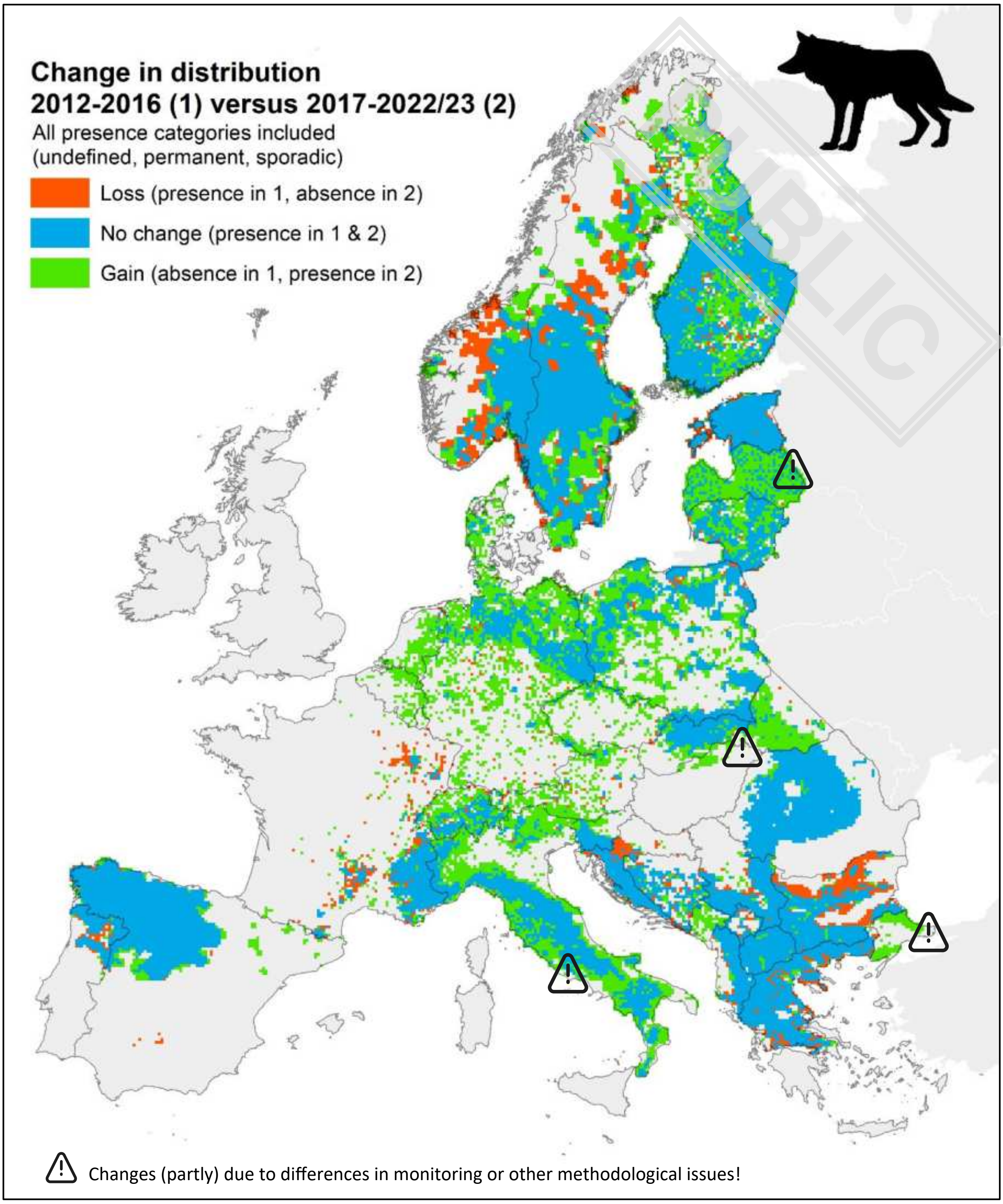

Fig. 12: Changes in wolf distribution 2012-2016 versus 2017-2022/23.

3.1.4. Wolverine

Overview of main distribution monitoring methods

The most important monitoring methods for determining the distribution of wolverines are dead animals, non-invasive genetics (scats), camera traps, active snow tracking, natal-den (family group) detection, damage statistics in Norway and Sweden, and confirmed wolverine signs and family group detection in Finland (Table 10).

Table 10: Main distribution monitoring method for the wolverine.

Country	Range monitoring method											
	Dead animals	Non-invasive genetics	Camera traps	GPS tracking	Active snow tracking	Howling surveys	Family groups	SCALP C2	Damage statistics	SCALP C3	Quest. & interviews	Past presence	Other*
Wolverine													
Finland		<10%					25-50%	>75%					
Sweden & Norway	>75%	>75%	>75%		>75%		>75%		>75%				

Overview of main mapping methods

Norway and Sweden use identical distribution monitoring and mapping methods, with buffers around confirmed wolverine signs of reproduction to define the area of permanent presence. In Finland, unbuffered confirmed signs of wolverines are used to define the range. Information on reproduction is not systematically collected, so that the distinction between permanent and sporadic presence, which was done in the 2012-2016 mapping, is no longer possible for Finland (Table 11).

Table 11: Mapping details for wolverine in Europe.

Country / Region	FINAL_time	Spatial scale	% Known range monitored		Large carnivore signs used	Definition of gridcells based on	Scale of data quality information	Presence categorisation based on	Method change	Range trend estimate since 2012-2016	
			Active	Passiv						Trend	Assessment
Wolverine											
	2022/23	Entire known range	100	80	C 1 & C2	Confirmed presence signs overlaid with the 10 x 10 grid	Cell level	No distinction was made	No	Increasing	Real
Sweden	2021	Entire known range	100	100	C1 & C2	Buffered confirmed presence signs overlaid with the 10 x 10 grid	Country level	Reproduction only	No	No obvious change	Real
	2021	Entire known range	100	100	C1 & C2	Buffered confirmed presence signs overlaid with the 10 x 10 grid	Country level	Reproduction only	No	No obvious change	Real

Current wolverine distribution in Europe 2017-2022/23

The wolverine is only found in the 3 northernmost countries, Norway, Sweden, and Finland. The total distribution area of wolverines in Europe currently covers 745,00 km^2, which is a 4% increase in range since 2016 (Fig. 13, Table 12).

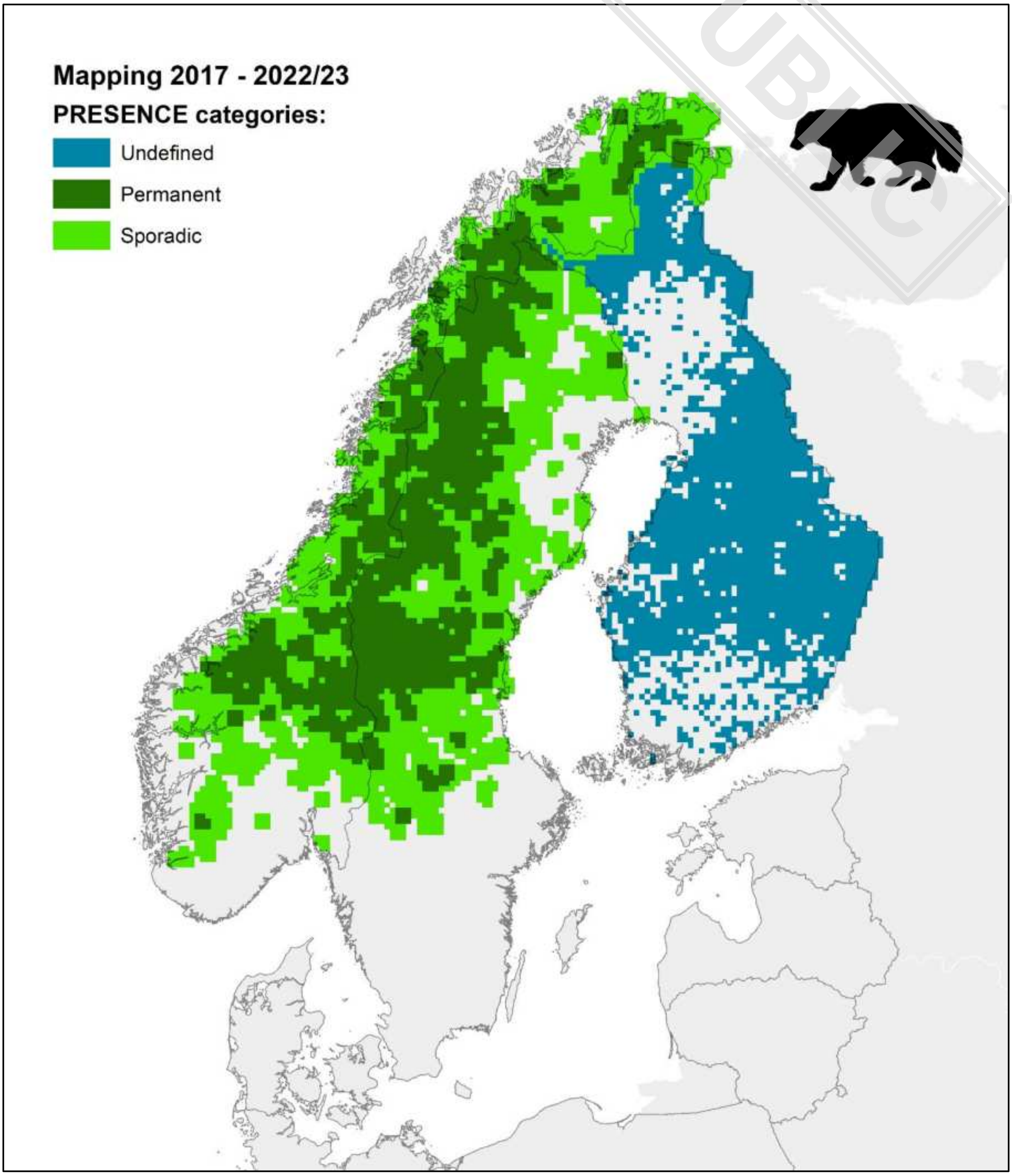

Fig. 13: Wolverine distribution in Europe for the period 2017-2022/23.

The distribution of the wolverine is entirely based on confirmed presence signs in Finland and on confirmed and buffered presence signs in Norway and Sweden. For Finland the distinction between permanent and sporadic distribution is no longer possible as the monitoring of den sites is greatly reduced and therefore no longer effectively represents the permanent range (Fig. 14).

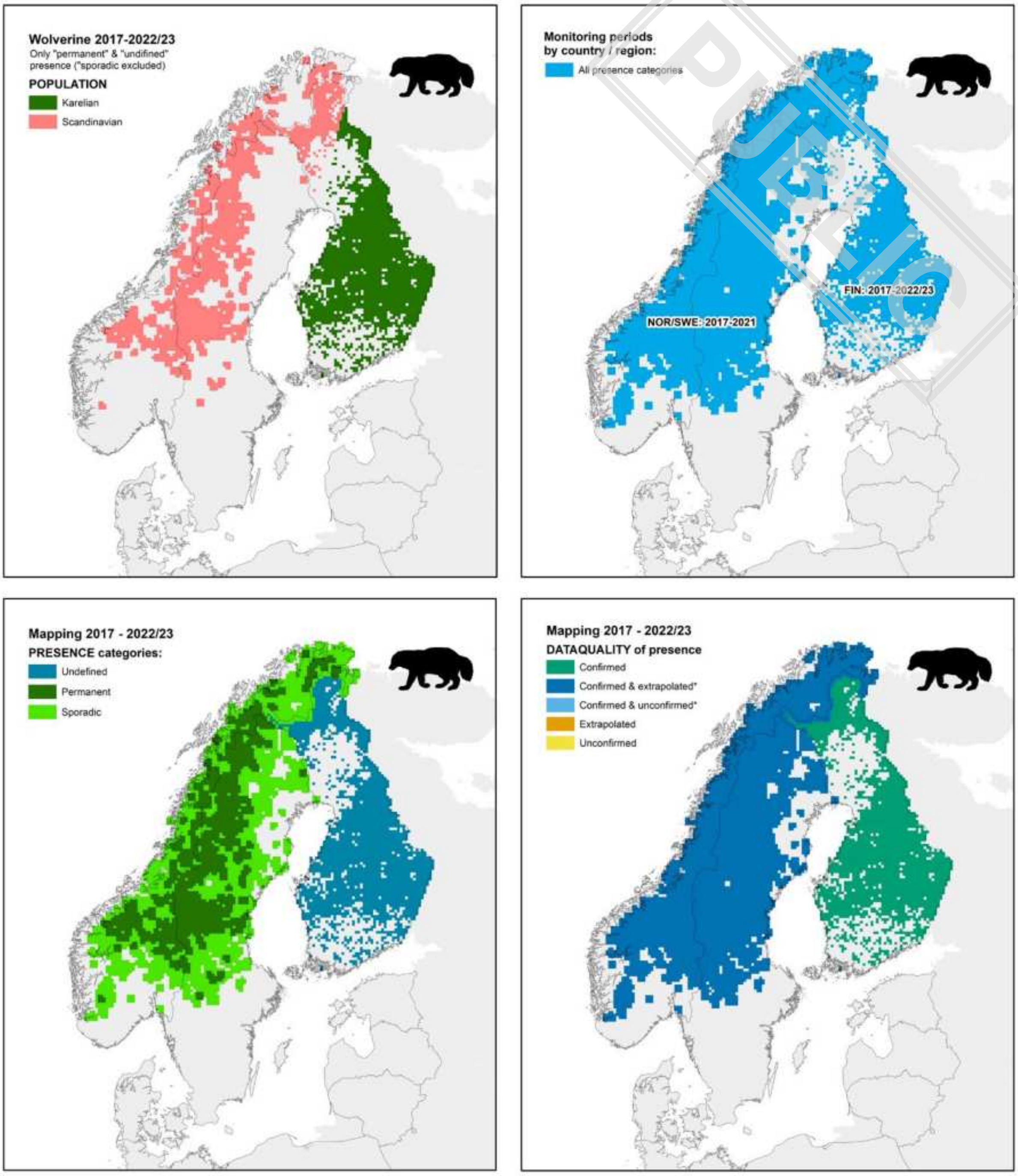

*Fig. 14: Wolverine populations, countries/regions with monitoring periods deviating from the 2017-2022/23 period, wolverine distribution, and underlaying data quality. *DATAQUAL: Mixed layer of confirmed and extrapolated or unconfirmed cells, where no separation at cell level was possible.*

Change in wolverine distribution 2012-2016 versus 2017-2022/23

Wolverine distribution seems to have mainly remained stable. Looking at individual populations (Fig. 14 & 15, Table 12):

- The **Scandinavian population's** distribution has remained largely stable with slightly more losses than gains along the fringes. The area has been self-reported as showing no obvious change by both Norway and Sweden.
- The **Karelian population's** distribution also seems to be expanding in the south and north-west. The area has been self-reported as increasing by Finland.

Table 12: Changes in wolverine distribution in Europe since 2016, expressed as number of 10 x 10 km cells.

Population	N cells in 2016				N cells in 2022				Balance (%)			
	Permanent	Sporadic	Undefined	Total	Permanent	Sporadic	Undefined	Total	Permanent	Sporadic	Undefined	Total
Karelian	1,073	784		1,857			2,343	2,343	NA	NA	NA	26.2
Scandinavian	2,350	2,957		5,307	2,200	2,907		5,107	-6.4	-1.7		-3.8
Total	**3,423**	**3,741**		**7,164**	**2,200**	**2,907**	**2,343**	**7,450**	**-35.7**	**-22.3**		**4.0**

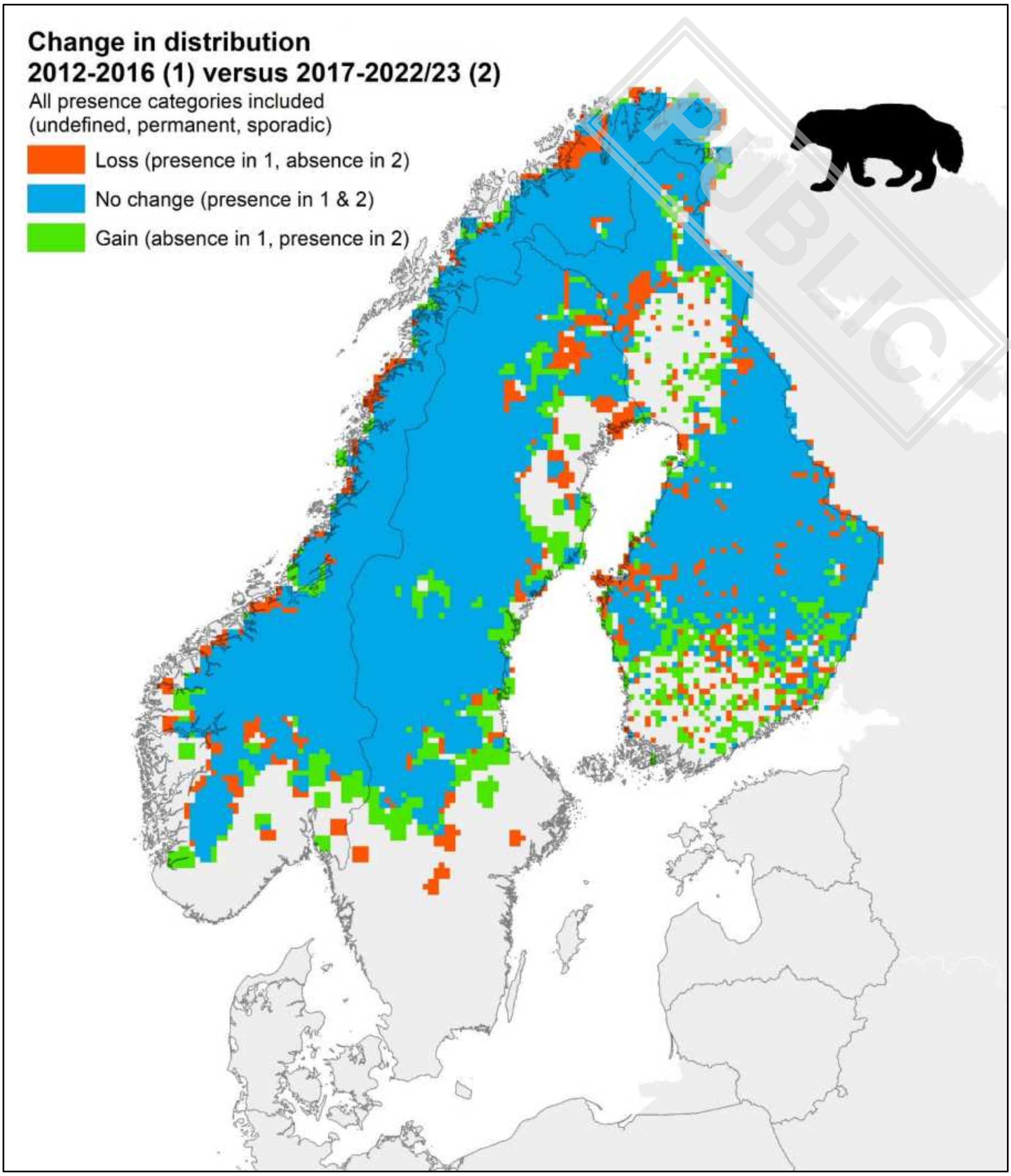

Fig. 15: Changes in wolverine distribution 2012-2016 versus 2017-2022/23

3.1.5. Golden jackal

Overview of main distribution monitoring methods

Overall, golden jackals are rarely subject to regular, intensive and active monitoring at regional or national scales. In the core of the species range, monitoring is primarily based on recorded dead animals, mainly from hunting bags. Howling surveys, camera traps, citizen-science based data and C3 records are especially prevalent in countries at the expansion front. Methods under "other" (Table 13) are ground- and aerial-based observations using thermal imaging in Greece.

Table 13: Main distribution monitoring method for the golden jackal.

Country	Range monitoring method												
	Dead animals	Non-invasive genetics	Camera traps	GPS tracking	Active snow tracking	Howling surveys	Family groups	SCALP C2	Damage statistics	SCALP C3	Quest. & interviews	Past presence	Other*
Golden jackal													
Albania	<10%		10-15%			10-25%		10-25%				<10%	
Austria	10-25%	10-25%	25-50%		<10%	<10%			10%				
Bosnia & Herzegovina	>75%		10-25%			>75%					<10%		
Bulgaria	>75%		25-50%					<10%				10-25%	
Croatia	>75%												
Czech Republic	primarily opportunistic												
Denmark	10-25%	25-50%	25-50%										
Estonia	25-50%		10-25%			10-25%							
Finland	No dedicated monitoring; entirely opportunistic												
France	No dedicated monitoring; entirely opportunistic												
Germany		<10%	>75%				<10%	<10%					
Greece	<10%		<10%			<10%							>75%
Hungary	>75%												
Italy	<10%		25-50%		<10%	25-50%					<10%		
Kosovo					<10%								
Latvia	No dedicated monitoring; entirely opportunistic												
Lithuania	No dedicated monitoring; entirely opportunistic												
Montenegro												>75%	
North Macedonia	10-25%		10-25%			25-50%					10-25%	<10%	
Norway	No dedicated monitoring; entirely opportunistic												
Poland	Little dedicated monitoring; largely opportunistic												
Romania										>75%			
Serbia	50-75%				<10%	<10%							
Slovakia	>75%	<10%	>75%	>75%	<10%	10-20%	>75%	>75%		>75%			
Slovenia	>75%												
Spain	No dedicated monitoring; entirely opportunistic												
Switzerland	No dedicated monitoring; entirely opportunistic												
The Netherlands	No dedicated monitoring; entirely opportunistic												
Turkey												>75%	

Overview of main mapping methods

For most countries, distributional data for golden jackals was available between 2017 and 2022/23, and in some rare cases (e.g., Kosovo* and Spain) until 2024. Some countries either used fragmented but recent data (Romania) or older records (Montenegro, North Macedonia and Turkey; Fig. 16). Most of the presence data is based on confirmed (C1) records, but in some countries also C2 and C3 are reported. In Romania, jackal presence data came in the form of unconfirmed jackal population estimates (C3) at hunting ground level.

Most commonly, signs of presence were intersected directly with the 10 x 10 km grid and are reported or later intersected with cell level. In nine countries (Bosnia and Herzegovina, Bulgaria, Croatia, Estonia, Hungary, Lithuania, Romania, Slovakia, Slovenia), however, presence signs were aggregated at the hunting ground level, which were then intersected with the 10x10 km grid. In these countries, the cells overlapping with hunting grounds where jackals were detected were assigned presence (the threshold of overlap was unknown in most cases; 5% was used for Croatia and Romania and 10% for Slovakia and Ukraine). The distribution of jackals may have therefore been overestimated in these countries due to the spatial resolution of the input data. Three countries (Albania, Latvia and North Macedonia) relied on buffers around jackal signs, and extrapolated presence and modelling was used in Albania.

The distinction between permanent and sporadic presence was primarily made based on the re-occurrence of golden jackal signs in the core distribution, or confirmed reproduction on the expansion front, where potential dispersers were assigned sporadic presence. Some countries on the expansion front (Austria, Denmark, Latvia and Poland) largely classified jackal presence as undefined; however, given the observed pattern of jackal expansion, it is likely that most recorded signs are associated with dispersing individuals i.e., "sporadic presence". In some countries within the core of the distribution (Greece, Montenegro, Romania, Serbia, Slovenia and Turkey), the data was either too fragmented or too infrequent for making a reasonable distinction between permanent and sporadic presence; the cells were therefore classified as “undefined” (Table 14).

Table 14: Mapping details for the golden jackal in Europe.

Country / Region	FINAL_time	Spatial scale	% Known range monitored		Large carnivore signs used	Definition of gridcells based on	Scale of data quality information	Presence categorisation based on	Method change	Range trend estimate since 2012-2016	
			Active	Passive						Trend	Assessment
Golden jackal											
Albania	2017-2023	Entire known range	15	10	C1, C2 & C3	Presence is based on buffered confirmed presence signs which were overlaid with the 10 x 10 grid	Cell level and buffer of 2km	Re-occuring presence, and new records as	Yes (more howling surveys)	Increasing	Real & Method change
Austria	2017-2023	Entire known range	5	95	C1 & C3	Confirmed presence signs overlaid with the 10x10 grid	Cell level	confirmed presence	No	Increasing	Real
Bosnia & Herzegovina	2017-2023	Entire known range	65	35	C1 & C3	Confirmed presence signs overlaid with the 10x10 grid	Cell level	Re-occuring presence	No	Increasing	Real
Bulgaria	2017-2023	Entire known range	0	100	C1, C2 & C3	Confirmed presence signs overlaid with the 10x10 grid	Cell level	Re-occuring presence	No	No obvious change	
Croatia	2017-2023	Entire known range	0	95	C1	Presence based on larger areas (hunting grounds) with confirmed presence signs (hunted individuals) which were overlaid with the 10x10 grid	Cell level	Re-occuring presence	No	Increasing	Real
Czech Republic	2017-2022	Entire known range	<10	100	C1 & C3	Confirmed presence signs overlaid with the 10x10 grid	Cell level	Re-occuring presence, confirmed	Yes	Increasing	Real
Denmark	2016-2023	Entire known range	0	100		Confirmed presence signs overlaid with the 10x10 grid	Cell level	Re-occuring presence	No	Increasing	Real
Estonia	2017-2023	Entire known range	100	100	C1 & C2	Confirmed presence signs overlaid with the 10x10 grid	Cell level	Re-occuring presence	No	Increasing	Real
Finland	2017-2023	Entire known range	0	100	C1 & C2	Confirmed presence signs overlaid with the 10x10 grid	Cell level	Confirmed reproduction	No	Increasing	Real
France	2017-2023	Entire known range	0	100	C1	Confirmed presence signs overlaid with the 10x10 grid	Cell level	Confirmed reproduction	No	Increasing	Real
Germany	2017-2023	Entire known range	<1	99	C1 (Hatlauf & Böcker, 2022)	Confirmed presence signs overlaid with the 10x10 grid	Cell level	Confirmed reproduction	Yes	Increasing	Real
Greece	2017-2024	Entire known range	60	60	C1 & C2	Confirmed presence signs overlaid with the 10x10 grid	Cell level	NA	Yes	Increasing	Real & Method change
Hungary	2017-2022	Entire known range	0	100	C1	Confirmed presence signs overlaid with the 10x10 grid	Cell level	Re-occuring presence	No	Increasing	Real
Italy	2016-2023	Entire known range	<10	100	C1, C2 & C3	Confirmed presence signs overlaid with the 10x10 grid	Cell level	Re-occuring presence, confirmed	Yes	Increasing	Real
Kosovo	2017-2024	Only in certain reference areas	20	NA	C1	Presence based on confirmed presence signs and modeling based on habitat suitability and/or proximity criteria which were overlaid with the 10x10 grid	Cell level	NA	Yes	Unknown	Unknown
Latvia	2018-2023	Entire known range	0	100	C1	Presence is based on the location of confirmed presence signs which were overlaid with the 10 x 10 grid	Cell level	NA	Yes	Likely increasing	Likely real
Lithuania	2018-2024	Entire known range	0	100	C1	Presence is based on the location of confirmed presence signs which were overlaid with the 10 x 10 grid	Cell level	NA	No	Likely increasing	Likely real
Montenegro	2012-2016	NA	NA	NA	NA	NA	NA	NA	NA	Unknown	Unknown
North Macedonia	2014-2023	Selected areas	30	40	C1, C2 & C3	Presence based on confirmed presence signs and modeling based on habitat suitability and/or proximity criteria which were overlaid with the 10x10 grid	Cell level	NA	Yes	Increasing	Real
Norway	2017-2023	Entire known range	0	100	C1	Confirmed presence signs overlaid with the 10x10 grid	Cell level	NA	No	Likely increasing	Likely real
Poland	2017-2023	Entire known range	0	100	C1 & C2	Confirmed presence signs overlaid with the 10x10 grid	Cell level	NA	No	Likely increasing	Likely real
Romania	2022-2023	Entire known range	0	100	C3	Presence based on larger areas (hunting grounds) with unconfirmed presence signs (estimated number of individuals by hunters) which were overlaid with the 10x10 grid	Cell level	NA	Yes	Increasing	Real & Method change
Serbia	2017-2023	Entire known range	60	40	C1	Presence is based on the location of confirmed presence signs which were overlaid with the 10 x 10 grid	Cell level	Re-occuring presence, confirmed	No	Increasing	Real
Slovakia	2017-2022	Entire known range	0	100	C1, C2 & C3	Presence is based on larger areas (e.g. hunting grounds) with confirmed presence signs (e.g. hunted individuals) which were overlaid with the 10 x 10 grid	Cell level	Re-occuring presence, confirmed reproduction	No (some)	Increasing	Real
Slovenia	2017-2023	Entire known range	0	100	C1	Presence is based on the location of confirmed presence signs which were overlaid with the 10 x 10 grid	Cell level		No	Increasing	Real
Spain	2017-2024	Entire known range	0	100	C1	Confirmed presence signs overlaid with the 10x10 grid	Cell level	Confirmed reproduction	No	Increasing	Real
Switzerland	2017-2023	Entire known range	0	100	C1, C2 & C3	Confirmed presence signs overlaid with the 10x10 grid	Cell level	Confirmed reproduction	No	Increasing	Real
The Netherlands	2017-2023	Entire known range	0	100	C1	Presence is based on the location of confirmed presence signs which were overlaid with the 10 x 10 grid	Cell level	Re-occuring presence, confirmed	Yes	Increasing	Real
Turkey	2004-2013	Entire known range	0	100	C1 & C2	Presence in part of the range is based on unconfirmed presence signs, or assumed presence based on interviews, questionnaires, and media reports, or documented past presence (this past presence cannot be older than from 2010)	Cell level	No distinction	No	No obvious change	Method change

Current golden jackal distribution in Europe

The golden jackal is currently found in 29 of the 34 European countries/regions covered by this report. The species is thought to be absent only in Belgium, Luxembourg, Portugal, and Sweden. The total range encompassed by the species in Europe currently covers approximately 765,000 km², which is a 46% increase of its distribution since 2016 (Fig. 16, Table 15).

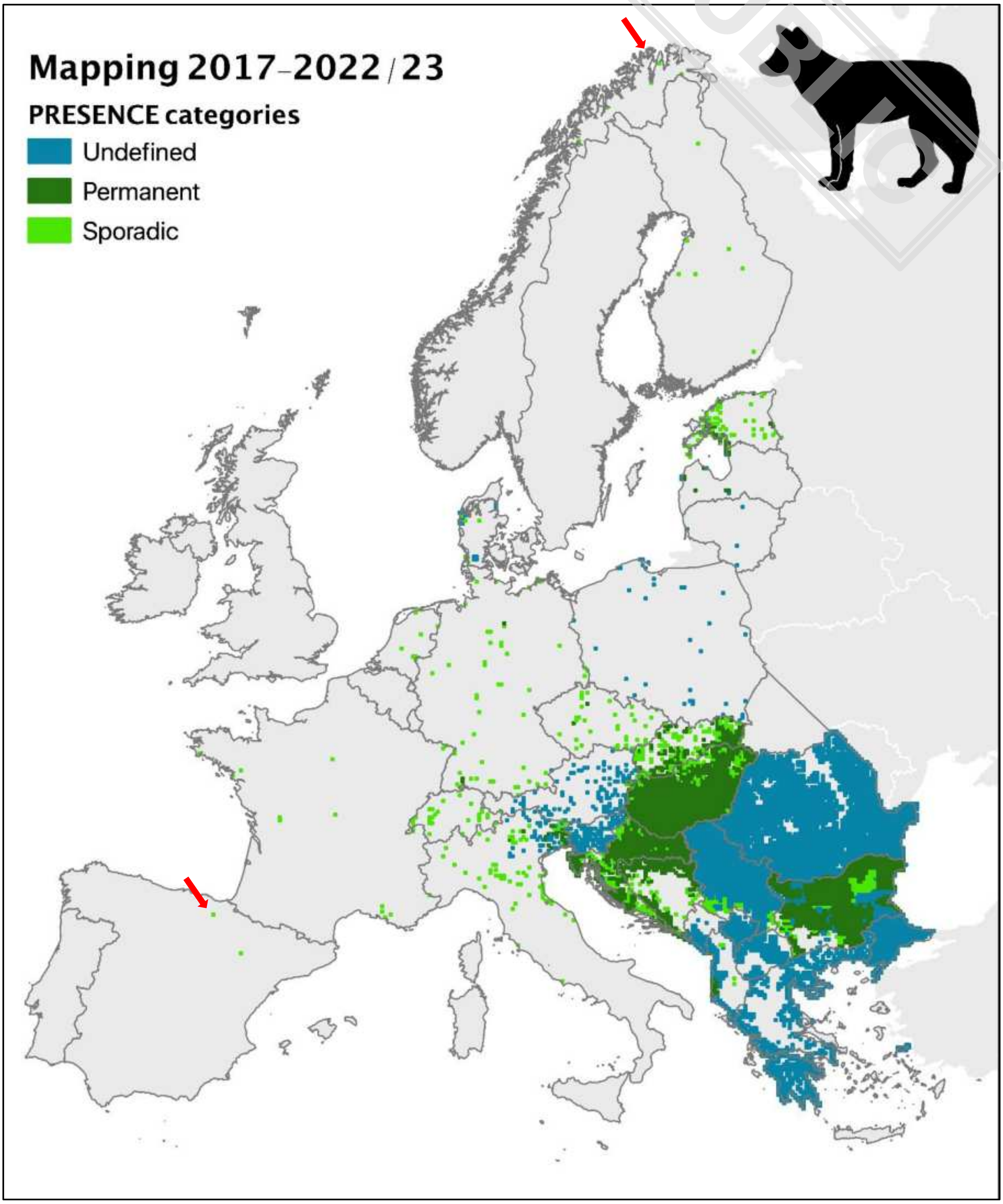

Fig. 16: Golden jackal distribution in Europe for the period 2017-2022/23. Note: Isolated cells away from the main population are slightly enlarged for better visibility and arrows point to Europe's northernmost and easternmost occurrence records.

Golden jackals are primarily distributed within a single interconnected Continental population. The (previous) 2012-2016 assessment distinguished between Continental and Adriatic populations (Ranc et al. 2022). Due to the large contact zone observed in the Dinaric and Balkan population range, we now treat these previously separated populations jointly as the Continental population. In contrast, two populations in Greece, on the island of Samos and on the Peloponnese peninsula, remain largely isolated (Fig. 17).

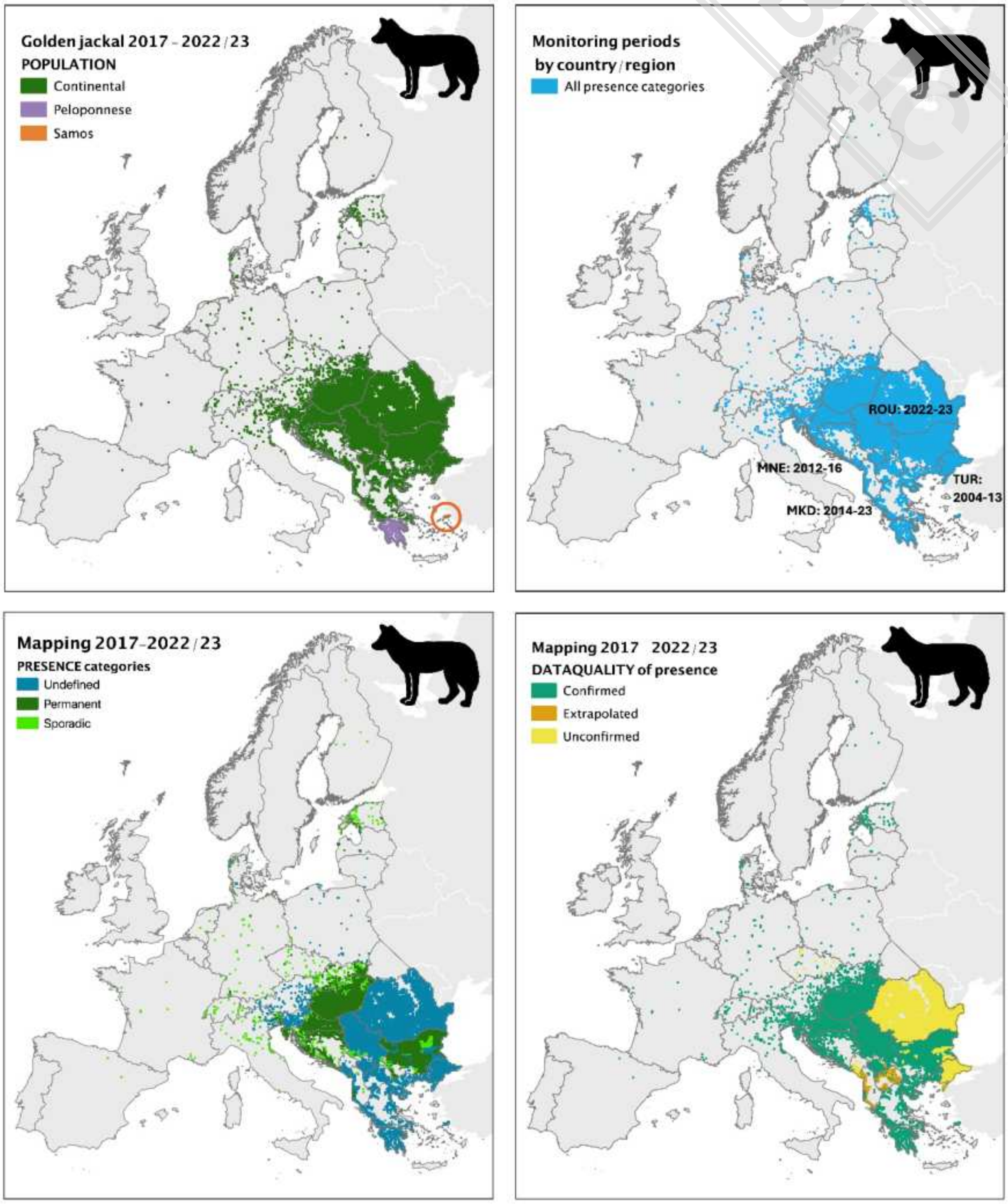

Fig. 17: Golden jackal populations, countries/regions with monitoring periods deviating from the 2017-2022/23 period, golden jackal distribution, and underlaying data quality

The present estimation of distribution does not cover Belarus, Moldova and Ukraine. Dispersing jackals are known to occur in Belarus while established populations seem to be present in both Moldova and Ukraine.

Change in golden jackal distribution 2012-2016 versus 2017-2022/23

The trend in golden jackal distribution area was estimated to be increasing in 21 countries. Two countries reported no obvious change. Due to infrequent or fragmented monitoring, the trend in jackal presence is unknown in six additional countries. No country/region reported a decreasing area of distribution (Table 15).

To quantitatively compare these changes, we reclassified the (previous) 2012-2016 distribution such that “expert-based absences” were considered as “absences”, “expert-based presences” as “undefined presence”, and assigned “Adriatic”, “vagrants” and “NA” to the “Continental” population. Overall, the golden jackal range seems to have significantly increased, although changes in monitoring methods, effort, or mapping have likely also had some regional influence (Fig. 18; Tab.15).

- The **Continental population’s** distribution is characterized by a significant expansion in the Dinaric mountains (Croatia and Slovenia), in Central Europe (especially Hungary and Serbia) and in the small nucleus in the Baltic States (Estonia and spreading into Latvia). Important (absolute) increases are noted in Greece and Romania, but these may be partly the result of a change in monitoring and mapping methods. Turkey was not covered in the previous assessment (2012-2016) other than the border areas, so gains are largely the result of a dedicated mapping for 2017-2022/23. No new data for Montenegro were available since the previous assessment so that changes in local jackal distribution are not possible to identify quantitatively. Long-distance dispersers are being noticed throughout western and northern Europe, with several countries (Finland, France, Norway and Spain) recording their first jackal presence during the 2017-2022/23 period (Hatlauf et al. 2021). In western and central Europe, particularly in Austria and the Czech Republic, there are both local losses and gains (but overall, largely gains), which likely reflect the ongoing jackal expansion process with a dynamic settlement/disappearance of dispersers.
- The distribution of the populations on the **Peloponnese** and **Samos Island** (both in Greece) appears relatively stable, although changes in monitoring makes the trend in the Peloponnese relatively unclear.

Table 15: Quantitative changes (km^2) in golden jackal distribution 2012-2016 versus 2017-2022/23, expressed as number of 10 x 10 km cells.

Population	km^2 in 2016				km^2 in 2022				Balance (%)			
	Permanent	Sporadic	Undefined	Total	Permanent	Sporadic	Undefined	Total	Permanent	Sporadic	Undefined	Total
Continental	319,918	93,212	52,130	**465,260**	250,431	71,326	424,377	**746,134**	-22	-23	714	60
Samos	480	0	0	**480**	0	0	480	**480**	NA	NA	NA	0
Peloponnese	5,100	495	15,777	**21,372**	0	0	18,363	**18,363**	NA	NA	NA	-14
Total	**325,498**	**93,707**	**67,907**	**487,112**	**250,431**	**71,326**	**443,220**	**764,977**	**-22**	**-23**	**714**	**46**

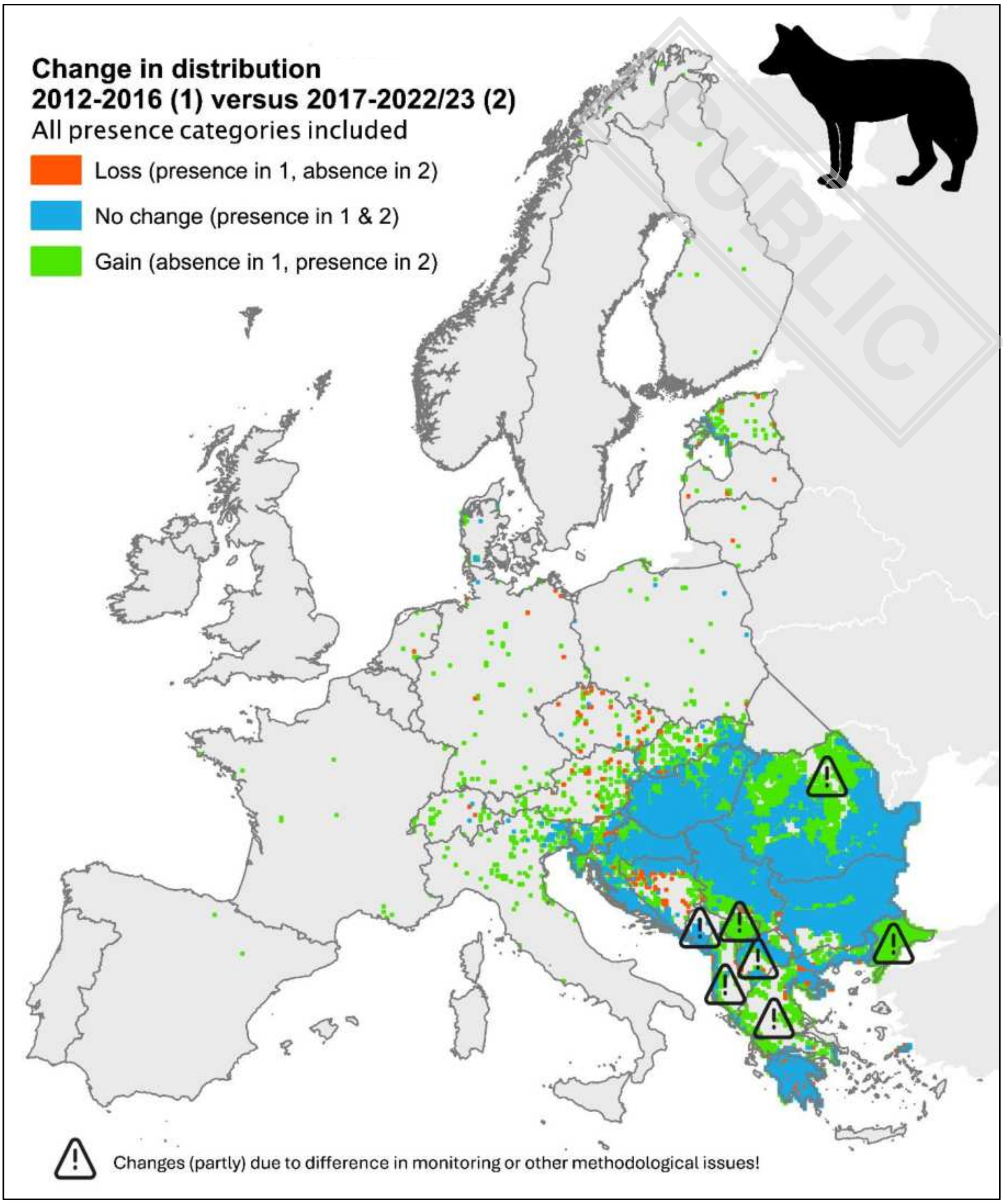

Fig. 18: Changes in golden jackal distribution 2012-2016 versus 2017-2022/23.

3.2. Population estimates

Although the data presented in this section represent the best available figures for each species and each country there is a wide diversity of methods in use, and a massive variation in the accuracy and precision of the numbers produced. For several countries and species no recent (France, Spain), or no reliable, population estimates were available (Kosovo*, Ukraine, Turkey) at all. Some are only based on expert assessments or informed guesses, whereas others result from more robust methodology.

Many populations are shared among several countries and individuals or social groups which have transboundary ranges and may be counted in more than one country. While monitoring and population estimates are harmonised among countries sharing borders for some species and populations, for many transboundary populations there is no correction for the number of individuals or social groups with transboundary ranges. While the number may be relatively small where populations just share short borders, the proportion of transboundary individuals can be very large, where populations are primarily found along international borders.

However, even when the same method is used, it can be used in many different ways. Camera trapping and the use of non-invasive DNA from scats / hairs are widely regarded as the gold standard methods for many species. But they can be used in different ways, either to add up the known individuals or produce statistical estimates of density with confidence intervals based on capture-recapture analyses. The area covered by surveys may vary, with different approaches to extrapolate to the rest of the distribution area. Different statistical approaches may produce different estimates. Different field methods also target different population metrics. Some survey all individuals, whereas others only survey certain parts of the populations, such as adults, or just document the presence of reproduction. Various conversion factors exist to allow conversion between the number of wolf packs and the number of wolves, for example. Unfortunately, there is as yet little standardisation of these conversion factors even within the different parts of the same population.

Surveys can also be conducted at different times of the year which can reflect very different population sizes, especially in areas where hunting is conducted. Methods also change and adapt over time, which makes comparisons with older data harder. A final challenge concerns the availability of data as we had several cases where data is known to be available, but is not accessible, or not sufficiently well documented to include.

There are encouraging signs of more sophisticated methods being used in more areas, although some regions, such as southeastern Europe with their large populations, suffer from a major underinvestment in monitoring activity. There are also areas where neighbouring countries make great efforts to standardise their field methods and analysis, such as the Alps, Scandinavia, or the Pyrenees. Such efforts need to be expanded so that methods within populations are harmonised.

Overall, it is important to treat all figures with a certain degree of caution, look for the bigger picture and refer to the overview tables on methods and data quality and go back to the original literature (many of which is listed in Appendix 6). Despite this caution we believe that they do reflect the general size and trend of large carnivore populations in Europe relatively well. We have gone to great lengths to make them as comparable as possible, and to make the underlying variation in methodology as transparent as possible. But the diversity is huge (also see Appendix 3 for a few random examples).

The question asked for compiling the main monitoring methods was rather complex as it was asking for "What is the main monitoring method to obtain population estimates and how much of the species range is approximately monitored with this method (% area)?" Hence selecting a method category, was meant to identify it as the main monitoring method and the % was meant to provide information what % of the population was monitored with this method. However, the question was not always interpreted this was and some categories/terms were not entirely clear. Hence the overview has to be interpreted with some caution and for more details the original literature in Appendix 6 should be consulted.

3.2.1. Brown bear

Brown bear population estimates in Europe are by now widely based on non-invasive genetic monitoring either determining the minimum number of individuals or calculating densities using capture-recapture methods, although observations/detection of females with cubs-of-the-year (coys), and harvest data, are still used in some countries (Table 16).

For details on country specific large carnivore population estimate methods and/or monitoring methods please see references in Appendix 6.

Main methods to estimate brown bear populations in Europe

Table 16: Main monitoring methods for bears in Europe. Min = minimum number, Repro = reproduction, CMR = capture-mark-recapture, Pres = presence, YoY = Young of the Year, Obs = observation; (Scats Repro and Howling surveys not relevant for bears)

Country/Region	Camera traps			Snow tracking				Non-invasive genetics		Scats	Howling surveys		YoY obs	Hunter data		Expert estimate	Other
	Min	Repro	CMR	Min	Repro	Natal dens	Track index	Min	CMR	Repro	Repro	Pres		Obs index	Harvest data		
	Brown bear																
Albania		25-50%						10-25%		10-25%							>75%
Austria								>75%									
Bosnia & Herzegovina	50-75%	10-25%						10-25%							25-50%	<10%	
Bulgaria															0	0	25-75%
Croatia	10-25%	10-25%	<10%	<10%	<10%		<10%	>75%	>75%				50-75%	50-75%	>75%	25-50%	
Czech Republic	>75%	>75%		10-25%			50-75%	<10%									
Estonia		>75%											>75%				
Finland	<10%	10-25%	<10%	<10%	<10%	<10%	<10%	<10%	<10%				>75%	<10%	25-50%		
France								>75%	>75%								
Germany	>75%							>75%									
Greece	>75%	50-75%						>75%	25-50%				25-50%				
Hungary	25-50%	<10%	<10%	10-25%	<10%	<10%	<10%	10-25%	<10%				<10%	<10%		50-75%	10-25%
Italy - Alps								>75%	>75%								
Italy - Apennine								10-25%									
Kosovo*	no population estimates available																
Latvia	>75%	>75%						>75%					>75%				>75%
Lithuania																	
Montenegro																	
North Macedonia								10-25%								50-75%	
Norway								>75%	>75%								
Poland									>75%								
Romania				>75%				<10%					10-25%				
Serbia	50-75%	<10%						<10%					<10%				<10%
Slovakia	>75%	>75%	<10%	>75%	>75%	<10%	<10%	>75%	>75%	<10%			>75%	<10%	<10%	<10%	
Slovenia	<10%	<10%	<10%	<10%	<10%	<10%	<10%		>75%	<10%			<10%	<10%	>75%	<10%	
Spain									>75%				10-25%				
Sweden								>75%	>75%					>75%	>75%		
Switzerland	10-25%			10-25%				10-25%								10-25%	
Ukraine - Carpathians	Population estimates are currently based on counts from protected areas, hunting grounds, and forest units, which are not corrected for double counts.																

Most recent population estimates and changes in population estimates for brown bears in Europe

The bear population in 2023 is estimated at around 20,500 individuals. The increase since 2016 (see https://www.lcie.org/Large-carnivores/Brown-bear) is largely due to an increase in the Baltic, Carpathian, and Karelian populations (Table 17).

However, population estimates in the Dinaric-Pindos, Carpathian, and Eastern Balkan population are still in part based on older estimates, expert opinion, and flawed methods (see Table 18). The population estimate for the Cantabrian mountains is from 2000, but the population is increasing and in 2023 there may potentially be as many as 400 bears. The population estimate for the small and isolated Central Apennine bear population has not been updated since 2014 and the actual trend for this isolated population is therefore unknown.

Table 17: Population trend of brown bears in Europe since the last update in 2016. ***Unknown** = it is not known what number or % of animals are counted in more than one country,* ***Excluded** = coordinated monitoring excluded double counting.*

Population	Countries	Estimate 2012-2016	Estimate 2017-2023	Trans-boundary double counts*	Trend	Comment
Alpine	Italy, Switzerland, Austria, Slovenia	49-69	100	Few	↑	
Baltic	Estonia, Latvia	700	1,090	Excluded for females with coys*	↑	
Cantabrian	Spain	321-335	324	No border	↑	Change in methods since 2016. The reported estimate is from 2020. The population is clearly increasing and as of 2023, may be around 400 bears.
Carpathian	Romania, Poland, Slovakia, Serbia, Ukraine, Hungary, Czech Republic	7,630	9,000	Considerable in some cases**	↑	Population estimates partly contested, some expert estimates, no robust data from UKR
Central Apennine	Italy	45-69	50	No border	→	**No update since 2014!**
Dinaric-Pindos	Slovenia, Croatia, Bosnia & Herzegovina, Montenegro, North Macedonia, Albania, Serbia, Kosovo*, Greece	3,950	4,112	Excluded only for SVN / HRV; there are also some regional	→	Data partly older, includes expert opinion for RS

				trans-boundary initiatives		
Eastern Balkan	Bulgaria, Greece, Serbia	468-665	460	Unknown	→	Method flaws for estimates in BGR, includes expert opinion for SRB
Karelian	Norway, Finland	1,660	2,175	Unknown	↑	NOR did not report northern bears separately in 2022
Pyrenean	France, Spain	41***	86	Excluded	↑	Coordinated transboundary monitoring
Scandinavian	Norway, Sweden	2,825	3,000	Excluded	→	Coordinated transboundary monitoring
Total (c.)		**17,000 - 18,000**	**20,400**		↑	

**coys = cubs of the year;* **Double counts exist on the Pan-Carpathian level, as there are only now some attempts within the Carpathian Convention and some international projects to synchronise monitoring between countries (e.g., LECA project https://www.interreg-central.eu/projects/leca/); ***Estimate was corrected from faulty previous estimate (which was 30)

Table 18: Population estimates for the brown bear in Europe by country and population. Note: Country level population estimates may include double counting of transboundary individuals. Where population level estimates were available these were used for the sum here and in Table 17. For references see Appendix 6.

Bear population	Year(s)	Estimate (indiv.)	Uncertainty	Details	% of range monitored	Trend	Trend quality	Reference
Alpine		**100**						
Austria	2023	4	transboundary	min. average/year	100	No obvious trend	Real	https://baer-wolf-luchs.at/verbreitungskarten/baer-verbreitung
Germany	2022/23		sporadic (1)	minimum	NA	NA	NA	https://www.lfu.bayern.de/natur/wildtiermanagement_grosse_beutegreifer/baer/monitoring/index.htm
Italy	2023	98	95% CI 86-120	>1 year, DNA profile	100	Increasing	Real	https://grandicarnivori.provincia.tn.it/Large-Carnivores-Report
Switzerland	2017-2023		sporadic (0-2)	minimum	NA	NA	NA	https://www.kora.ch/de/arten/baer/verbreitung
Baltic		**1,090**						
Estonia	2023	960	minimum	females & coys x 10	100	Increasing	Real	https://keskkonnaportaal.ee/sites/default/files/2023-08/SEIREARUANNE_2023-fin.pdf
Latvia	2023	130	expert opinion	minimum	100	Increasing	Real	*In prep.*
Lithuania	2023		sporadic	no estimate	NA	NA	NA	No publications
Poland	2023		sporadic	minimum (0-1)	NA	NA	NA	Diserens et al. 2020
Cantabrian[1]		**324**						
Portugal	2023		sporadic (1)		NA	NA	NA	Media report
Spain - Cantabria East	2017	49	95% CI: 33.8-67.6	genetic CMR	100	Increasing	Real	López-Bao et al. 2020
Spain - Cantabria West	2019	275	95% CI: 222.5-338.3	genetic CMR	100	Increasing	Real	López-Bao et al. 2021
Carpathian		**9,000**						
Czech Republic	2023	2	sporadic (1-3)	minimum	95	NA	NA	No publication for 2023
Hungary	2023	12	10-15	expert opinion	70	Increasing	Real	No publication for 2023
Poland -Tatra	2017	55	95% CI: 45-79	first genetic CMR	100	Unknown	no previous baseline	Konopiński et al. 2018
Poland - Podkarpackie	2014-2015	72	95% CI: 45.2–115.5	first genetic CMR	100	Unknown	no previous baseline	Berezowska-Cnota et al. 2023
Romania	2018	6,825	6,450-7,200[2]	range	70	No obvious trend	Real	Romania's Habitats Directive Report Art. 17, 2019; National Action Plan, 2018
Serbia	2023	12	10-14	expert estimate	60	Increasing	Real	No publications
Slovakia	2023	2,000	1,900-2,100[3]	extrapolated	100	Increasing	Real	Rigg, R. unpubl. data 2024
Ukraine - Carpathian	2019	???	Uncorrected counts: 375	Population estimates are currently based on counts from protected areas, hunting grounds, and forest units, which are not corrected for double counts.				Cherepanyn et al. 2023
Central Apennine		**50**						
Italy	2014	50	range: 45-69	genetic CMR	22	No obvious change	Unknown	Ciucci et al. 2015
Dinaric-Pindos		**4,112**						
Albania	2021	200	range: 190-210		50	No obvious change	Real	Skrbinsek et al. 2022
Bosnia and Herzegovina	2017-2023	950	SD: 900-1,000		75	Increasing	Real	Zubić et al. 2023
Croatia	2018[4]	937	95%CI: 846-1072	genetic CMR incl. coys	100	No obvious change	Real	Huber et al. 2019, Skrbinšek et al. 2017
Greece	2017; 2021; 2022; 2023[5]	600	range: 550-650	genetics	90	Increasing	Real	Pylidis et al. 2021, Tsalazidou-Founta et al. 2022
Kosovo*				No population estimate available				
Montenegro				No population estimate available				
North Macedonia	2020	325	range: 300-350	Relative Abundance Index	60	Increasing	Unknown[6]	Gonev 2022 unpubl. MES Report
Serbia	2023	110	range: 100-120	expert estimate	75	Increasing	Real	No publications
Slovenia	2024	990	range: 810-1,000	genetic & mortality	99	Increasing	Real	Jerina 2024, Jerina & Ordiz 2021
East Balkan		**459**						
Bulgaria	2021	353	unknown, minimum	official data	48	Decreasing	Unknown	Serbezov & Spassov 2023, Ministry of Environment and Waters. 2023
Greece	2020; 2021; 2022	100	range	genetics	85	Increasing	Real	Pylidis et al. 2021, Tsalazidou-Founta et al. 2022
Serbia	2023	6	4-8	expert estimate	50	Increasing	Real	No publications
Karelian		**2,175**						
Finland	2023	2,175	2,100–2,250	females & coys x 10	100	Fluctating	Real	Heikkinen et al. 2023
Pyrenean		**86**						
France, Spain & Andorra	2023	86	95% CI: 82-92	genetic & PCRD	100	Increasing		Vanpé et al. 2022, Sentilles et al. 2023
Scandinavian		**3,002**						
Norway[7]	2023	178	identified	genetic	100	Increasing	Real	Brøseth et al. 2024
Sweden	2022	2,824	2,587-3,080 (post-harvest)	genetic	100	Fluctating	Real	Åsbrink et al. 2023
Total		**20,398**						

[1]In 2020, the regional government of Castilla y Leon and the three other Cantabrian regional administrations conducted another genetic survey. The results were: 370 bears in total (250 in the western and 120 in the eastern subpopulations). However, these results are inconsistent with the surveys of females with cubs of the year conducted during the past 30 years and with the previous genetic surveys. In addition, no publications or technical reports are available, only a press release from the regional government of Castilla y León: https://comunicacion.jcyl.es/web/jcyl/Comunicacion/es/Plantilla100Detalle/1281372051501/NotaPrensa/1285242547188/Comunicacion

[2]Offical numbers reported to the EU in 2019. There is a lot of discussion around the population size and its trend. A national survey based on genetic sampling is ongoing. The objective is to get a robust minimum number of bears; the first results should be available in 2025.

[3]A national population estimate was calculated by extrapolating from a 2013/14 genetic CMR estimate using the long-term population growth rate of 4.5% that was calculated in Rigg & Adamec (2007). There has been a more recent genetic CMR estimate, announced in 2023, but its reliability has been widely questioned so we did not use it.

[4]Samples collected in 2015

[5]Data on species population and conservation status are being updated in the frame of the current 6-year reporting cycle to the EC in compliance with article 17 of the Habitats Directive 92/43. Results will be available by beginning of 2026.

[6]Trend based on Relative Abundance Index in a reference area (Mavrovo National Park) only.

[7]Norway reported only at the national level; the northernmost bears in Finnmark county (likely ~35) should have been added to the Karelian population.

3.2.2. Eurasian lynx

Main methods to estimate Eurasian lynx populations in Europe

The main method to obtain population estimates of lynx in Europe is camera trapping, especially to detect reproduction. Detection of reproduction is also widely done via snow tracking in the Nordic countries. However, expert opinion still plays a significant role in Bulgaria (where there are only single dispersers), Latvia, Lithuania, Romania, and Poland (Table 19).

Table 19: Main monitoring methods for Eurasian lynx in Europe. Min = minimum number, Repro = reproduction, CMR = capture-mark-recapture, Pres = presence, YoY = Young of the Year, Obs = observation, Scats Repro = heaped occurrence of scats, suggesting a den site (this category was poorly explained).

Country/Region	Camera traps			Snow tracking				Non-invasive genetics		Scats	Vocalisation surveys		YoY obs	Hunter data		Expert estimate	Other
	Min	Repro	CMR	Min	Repro	Natal dens	Track index	Min	CMR	Repro	Repro	Pres		Obs index	Harvest data		
	Eurasian lynx																
Albania	>75%		>75%					25-50%									
Austria - Alps	25-50%	<10%	<10%	<10%	<10%	<10%	<10%	<10%	<10%	<10%	<10%	<10%	<10%	10-25%	<10%	<10%	
Austria - BBA	>75%	>75%															
Belgium	25-50%							>75%									
Bosnia & Herzegovina	50-75%	25-50%	10-25%	25-50%				<10%								<10%	
Bulgaria	<10%															>75%	
Croatia	>75%	>75%	>75%	10-25%	10-25%	10-25%	10-25%	25-50%	25-50%				50-75%	10-25%	<10%		
Czech Republic	>75%	>75%	25-50%	25-50%	10-25%		10-25%	25-50%									
Estonia		25-50%			>75%	>75%							>75%				
Finland													>75%				
France	>75%	>75%						0					>75%				
Germany	>75%	>75%						10-25%									
Hungary	50-75%	<10%	<10%	10-25%	<10%	<10%	<10%	10-25%	<10%	10-25%	<10%	<10%	<10%	<10%		50-75%	10-25%
Italy	>75%	>75%	<10%	10-25%	10-25%	<10%	<10%	<10%	<10%	<10%	<10%	<10%	>75%	<10%	<10%	25-50%	
Kosovo*	no population estimates available																
Latvia	<10%	<10%						<10%							>75%	>75%	
Lithuania																>75%	
Montenegro																	
North Macedonia	25-50%	<10%	25-50%													25-50%	
Poland	<10%	<10%		10-25%	10-25%		<10%	<10%					10-25%			>75%	
Romania	<10%	<10%	<10%	>75%			>75%	<10%					0	>75%		>75%	
Serbia	50-75%			<10%												<10%	
Slovakia	>75%	>75%	10-25%	>75%	>75%	<10%	<10%	<10%	<10%	<10%	<10%	<10%	>75%	<10%	<10%	<10%	
Slovenia*	>75%	>75%	>75%														
Sweden & Norway		25-50%			>75%												
Switzerland	>75%		50-75%														>75%
Ukraine - Carpathians	Population estimates are currently based on counts from protected areas, hunting grounds, and forest units, which are not corrected for double counts.																

Most recent population estimates for the Eurasian lynx in Europe

The Eurasian lynx population in 2023 is estimated at around 9,000 individuals and thus is in the same magnitude than in 2016 (see https://www.lcie.org/Large-carnivores/Brown-bear; Table 19).

Most of the large populations are stable (Karelian, Scandinavian populations) or are increasing slowly (Baltic, Carpathian population). However, population monitoring does not allow for robust population estimates or trend analyses in most of the range countries and monitoring methods and population estimation methods have changed (Table 21). The Dinaric population seems to have started increasing after 2019, reflecting the success of the extensive population reinforcement in recent years (Fležar et al. 2024). The Bohemian-Bavarian-Austrian seems to also slowly increase and expand. The Harz population is clearly increasing, which is in line with the documented range gains (see section 2.5). The Vosges-Palatinian population remains very small, despite additional re-introduction attempts (Port et al. 2024). The remnant population of the Critically Endangered Balkan lynx remains small and isolated and in urgent need of conservation actions (Melovski et al. 2021); Table 20).

Lynx population monitoring is coordinated for most populations, e.g. the Alpine population is coordinated under the Status and Conservation of the Alpine Lynx (SCALP) project and started to include distribution maps for adjacent populations. However, the unit counted – all individuals versus independents individuals (excluding dependent kittens), versus family groups – differs among countries (Table 21) and double counts in transboundary populations are not always excluded. Although family group counts were converted to individuals for the population estimates, it was not necessarily specified if “individuals” included kittens (all individuals) or only referred to independent individuals (Table 21). The Scandinavian population is monitored jointly by Norway and Sweden but lacks coordination with the adjacent Karelian population in Finland. The Baltic and Carpathian populations lack coordinated monitoring.

Table 20: Population trend of Eurasian lynx in Europe since the last update in 2016.

Population	**Countries**	**Estimate 2012-2016**	**Estimate 2017-2023**	**Trans-boundary double counts***	**Trend**	**Comment**
Alpine	Switzerland, Slovenia, Italy, Austria, France	163	255	Excluded (SCALP)	↑	Data from FRA missing; Reintroduction to the Julian Alps took place in 2021-2023
Balkan	Albania, Kosovo*, Montenegro, North Macedonia, Serbia	20 - 40	34	Likely 2-4 shared between ALB, XKX*, MKD	→	
Baltic	Estonia, Latvia, Lithuania, Poland	1,200 - 1,500	1,555	Unknown	↑	Pomeranian occurrence now listed separately
Bohemian-Bavarian-Austrian	Czech Republic, Germany, Austria	60-80	135	Excluded	↑	
Carpathian	Romania, Slovakia, Poland, Ukraine, Czech Republic, Hungary,	2,100-2,400	2,687	Unknown	↑	Robust data from UKR missing (in 2016: 330 lynx were assumed)

	Serbia, Bulgaria					
Dinaric	Slovenia, Croatia, Bosnia & Herzegovina	130	193	Excluded	↑	Population reinforcement took place in 2019-2023
Harz	Germany	46	113	No border	↑	
Jura	[France], Switzerland	140	>69	Excluded (SCALP) - data from FRA missing	??	Likely increasing
Karelian	Finland	2,500	2,483	Only 1 country	→	Double counts between populations possible
Scandinavian	Norway, Sweden	1,300 - 1,800	1,820	Excluded	↑	
Vosges-Palatinian	France, Germany	1 - 3	>12	Excluded (SCALP) - data from FRA missing	↑/→	Data for FRA missing. Reintroduction to the German part took place 2016 - 2020.
Pomeranian occurrence	Poland	NA	31	No border	NA	Formally included in the Baltic population
Black Forest-Swabian Jura occurrence	Germany	NA	5	Excluded within SCALP	NA	No reproduction, yet. Population reinforcement started in 2023.
Total (c.)		**8,000 - 9,000**	**9,400**		↑/→	Data from FRA missing

Unknown* = it is not known what number or % of animals are counted in more than one country, ***Excluded* = coordinated monitoring excluded double counting*

Table 21: Population estimates for the Eurasian lynx in Europe by country and population. Note: Country level population estimates may include double counting of transboundary individuals. Where population level estimates were available these were used for the sum here and in Table 20. For references see Appendix 6.

Lynx population	Year(s)	Estimate (indiv.)	Uncertainty	Details	% of range monitored	Trend	Trend quality	Reference
Alpine		**255**						
Austria	2023	20	17-23 minimum	independend individuals	80	Fluctuating	Real	unpubl. data
France	2023	???	no population estimates are done in France		100	NA	NA	unpubl. data
Germany	2022/23		sporadic (1-2)			NA	NA	unpubl. data
Italy	2023	7	range: 5-10			Increasing	Real	unpubl. data
Slovenia	2022/23	6		minimum count	90-100	Increasing	Real	Fležar et al. 2023 & 2024
Switzerland	2021	222	222 (+/- 9)	independent individuals	100	Increasing	Real	https://www.kora.ch/en/species/lynx/abundance
Balkan		**34**						
Albania	2023	7	range: 5-10	individuals	100	Decreasing	Real	Bego et al. 2022, Hoxha et al. 2023
Kosovo*	2023		few - no population estimates available			NA	NA	
Montenegro	2023		sporadic - no population estimates available			NA	NA	
North Macedonia	2023	25	20-30	independend individuals	50	No obvious change	Real	Melovski et al. 2013, Stojanov et al. 2020
Serbia	2023	2	sporadic, range: 1-3	expert estimate	20	NA	NA	unpubl. data
Baltic		**1,555**						
Estonia	2023	500	minimum number	86 females with kittens	100	Increasing	Real	Veeroja et al. 2023
Latvia	2021	700	range: 600-800 +/- 200	individuals	100	No obvious change	Real	Bagrade et al. 2016
Lithuania	2023	250	100-400	guestimate; no estmates available	100	Increasing	Unknown	Trend unknown due to lack of scientific data
Poland	2023	105	range: 58-151	density extrapolation to range	100	Fluctuating	Real	S. Nowak, R.W. Mysłajek, unpubl. data
Bohemian-Bavarian-Austrian		**135**						
Austria	2017/18-2022/23	22	mean 2017-2022	independant individuals	75-80	Fluctating	Real	Belotti et al. 2023
Czech Republic	2022/2023	81	minimum number	independant individuals	80	Increasing	Changed monito	unpubl. data
Germany	2022/23	50	minimum number	independent individuals	100	Increasing	Real	preliminary estimate - not yet published
Carpathian		**2,687**						
Bulgaria	2017-2023		sporadic	data not sufficient for estimates	1	NA	NA	Spassov et al. 2023
Czech Republic	2022/2023	11			95	No obvious change	Real	unpubl. data
Hungary	2022	12	range: 10-25	expert estimate	80	No obvious change	Monitoring chan	Annual reports by National Parks
Poland	2023	130	range: 72-188	density extrapolation to range	100	Increasing	Real	https://www.gov.pl/web/gios/poiis---monitoring-wilka-i-rysia
Romania	2018	2,250	2,100-2,400	minimum count	100	No obvious change	Unknown	Current monitoring not suitable to detect trend
Serbia	2023	50	range: 40-60	expert estimate	20	No obvious change	Real	unpubl. data
Slovakia	2022	234	min/man error: 155-325	density (1.11/100 km^2) extrapolation to range	100	No obvious change	Real	Rigg unpubl. Data, Appendix 3
Ukraine - Carpathians	2019	???	Uncorrected counts: 435	Population estimates are currently based on counts from protected areas, hunting grounds, and forest units, which are not corrected for double counts.				Cherepanyn et al. 2023
Dinaric		**193**						
Bosnia and Herzegovina	2017-2023	50	range: 40-60	individuals; incl. Expert estimate	85	Fluctating	Monitoring chan	Fležar et al. 2021, Trbojević et al. 2020
Croatia	2023	101	SD: 72-141		100	Increasing	Real	Gomerčić et al. 2023
Montenegro	2023		sporadic			NA	NA	
Slovenia	2022/23	42	95%CI: 30-60	SCR camera traps	90-100	Increasing	Real	Fležar et al. 2023 & 2024
Harz		**113**						
Germany	2022/23	113	minimum number	75 independent individuals, including 20 reproductions	100	Increasing	Real	preliminary estimate - not yet published
Jura		**69**						
Switzerland	2019	69	69 (+/- 5)	individuals	100	Stable	Real	https://www.kora.ch/en/species/lynx/abundance
France	2023		no population estimates are done in France		100	NA	NA	
Karelian		**2,483**						
Finland	2023	2,483	2,390-2,575	modelling based sighting of lynx and females with kittens	100	increasing	Real	Valtonen et al. 2023

Table 24 continues the next page

Table 24 continuation

Lynx population	Year(s)	Estimate (indiv.)	Uncertainty	Details	% of range monitored	Trend	Trend quality	Reference
Scandinavian		**1,820**						
Norway	2023	420	95% CI: 350 - 500	family groups on snow	100	No obvious change	Real	Frank & Tovmo. 2023
Sweden	2023	1,400	95% CI: 1,200 - 1,600	family groups on snow	100	No obvious change	Real	
Vosges-Palatinian		**12**						
France	2023	???	no population estimates are done in France		100	Increasing	Real	likely due to dispersal from Germany
Germany	2022/23	12	minimum number	individuals	100	Decrease	Real	no reproduction in 2022/23; Port et al. 2024
Pomeranian occurrence		**31**						
Poland	2021	31	61 released 2014-2020; no population estimate					Tracz et al. 2021, Skorupski et al. 2022
Black Forest-Swabian Jura occurence		**5**						
Germany	2023	5	minimum number	individuals	90-100	NA	NA	no reproduction in 2022/23; preliminary estimate - not yet published
Total		**9,392**						

3.2.3. Wolf

Main methods to estimate wolf populations in Europe

Monitoring methods for the wolf are most variable among countries and cover the entire spectrum of the most used methods. In addition, there is variation in 1) the units that are monitored (some countries only count packs and pairs, whereas others count individual wolves), 2) the conversion factors from packs to individuals, 3) the time period of the population estimates (e.g., prior to or at peak reproduction, pre- or post-hunting/culling; cumulative sampling over the entire year versus only during a specific time window etc.), and 4) estimation methods (e.g., density estimates versus minimum counts). Although the same is true for bear and lynx, the differences in the resulting population estimates are more extreme for wolves due to their higher reproductive potential (larger litter sizes and annual reproduction). Regionally, expert opinion remains important (e.g., Bulgaria, Latvia and North Macedonia (Table 22).

Table 22: Main monitoring methods for wolves in Europe. Min = minimum number, Repro = reproduction, CMR = capture-mark-recapture, Pres = presence, YoY = Young of the Year, Obs = observation, Scats Repro = heaped occurrence of scats, suggesting a Rendezvous site of wolves with pups (this category was poorly explained).

Country/Region	Camera traps			Snow tracking				Non-invasive genetics		Scats	Howling surveys		YoY obs	Hunter data		Expert estimate	Other
	Min	Repro	CMR	Min	Repro	Natal dens	Track index	Min	CMR	Repro	Repro	Pres		Obs index	Harvest data		
	Wolf																
Albania		10-25%									<10%	<10%					
Austria		<10%						>75%					<10%				
Belgium	25-50%							>75%					>75%				
Bosnia & Herzegovina	50-75%			10-25%								25-50%			10-25%	<10%	
Bulgaria				25-50%											0	50-75%	
Croatia	25-50%	25-50%	<10%	10-25%	10-25%	<10%	10-25%	50-75%	50-75%	<10%	10-25%	10-25%	10-25%	10-25%	<10%	<10%	<10%
Czech Republic	50-75%	50-75%		10-25%			25-50%	10-25%		<10%	<10%	<10%					
Denmark	25-50%	25-50%	<10%	<10%	<10%	<10%	<10%	25-50%	<10%	10-25%	<10%	<10%	25-50%	<10%	<10%	<10%	<10%
Estonia		50-75%		0	>75%		>75%				<10%		>75%				
Finland		<10%			<10%			>75%					>75%				
France								0	>75%								
Germany	>75%	>75%						>75%									
Greece	10-25%	10-25%						0	0	10-25%	10-25%						10-25% & >75%
Hungary	50-75%	25-50%	<10%	25-50%	<10%	<10%	10-25%	10-25%	<10%	10-25%	<10%	10-25%	25-50%	<10%		25-50%	<10%
Italy - Alps	>75%	>75%		>75%				>75%	>75%	>75%	>75%	0	>75%				
Italy - Peninsula									25-50%								
Kosovo*	no population estimates available																
Latvia															>75%	>75%	
Lithuania	10-25%			<10%											>75%		
Luxembourg	50-75%							50-75%									
Montenegro																	
North Macedonia																>75%	
Norway				>75%	>75%				50-75%								
Poland	10-25%	10-25%		<10%	<10%		<10%	10-25%	<10%	10-25%	<10%	<10%					>75%
Portugal	>75%	>75%						25-50%		>75%	>75%	>75%	>75%				
Romania	0& & 10-25%			>75%	<10%		>75%	<10%					0	>75%			
Serbia	10-25%	<10%								<10%			<10%		50-75%		
Slovakia	>75%	>75%	<10%	>75%	>75%	<10%	<10%	10-25%	10-25%	<10%	<10%	<10%	>75%	<10%	<10%	<10%	
Slovakia	>75%	>75%	<10%	>75%	>75%	<10%	<10%	10-25%	10-25%	<10%	<10%	<10%	>75%	<10%	<10%	<10%	
Slovenia									>75%								
Spain		50-75%						<10%		50-75%	>75%		>75%				
Sweden				50-75%	>75%				>75%								
Switzerland								>75%									
The Netherlands	>75%	>75%						>75%	>75%								
Türkey - Europe	<10%																
Ukraine - Carpathians	Population estimates are currently based on counts from protected areas, hunting grounds, and forest units, which are not corrected for double counts.																

Most recent population estimates for the wolf in Europe

The wolf population in 2023 is estimated at around 23,000 individuals, which is a 35% increase in population size since 2016 (see https://www.lcie.org/Largecarnivores/Wolf.aspx; Table 19). The trend is especially driven by the rapidly expanding Central European and Alpine populations. The estimate for the Carpathian population does not include Ukraine, and some of the estimates include expert estimates. The Dinaric Balkan population seems to be increasing, but in some areas, it is a struggle to get robust population estimates. The Alpine and Scandinavian populations are monitored at the transboundary level, and there is also increasing cooperation in the Dinaric-Balkan (e.g., via the Dinaric-Balkan-Pindos Large Carnivore Initiative) and the Central European (e.g., via the CEwolf consortium) populations. More specifically, monitoring of packs and pairs in the wider Alpine region is coordinated under the Wolf Alpine Group (WAG; Marucco et al. 2023) whereas the Scandinavian population is jointly monitored by Norway and Sweden. The CEwolf consortium, on the other hand, is working on a harmonised genetic monitoring of the Central European population to make genetic results comparable between different countries (Nowak et al. 2023). For several countries (e.g., Spain, Greece), only preliminary population estimates were available, as surveys and data analysis are still ongoing and for others there are currently no robust population estimates available (e.g., Ukraine, Turkey, Albania).

Table 23: Population trend of wolves in Europe since the last update in 2016.

Population	Countries	Estimate 2012-2016	Estimate 2017-2023	Trans-boundary double counts*	Trend	Comment
Alps and neighbouring areas	Italy, France, Switzerland, Germany, Austria, Slovenia	420 - 550	c. 2,000	Presence of double counts among countries in the estimate	↑	243 packs in the Alps were estimated in 2020-2021, with a coordinated effort among countries, avoiding double counts (Marucco et al. 2023)
Baltic	Estonia, Latvia, Lithuania, Poland	1,700 - 2,240	c. 3,000	Excluded between EE/LV; Unknown for rest	↑	
Carpathian	Slovakia, Czech Republic, Poland, Romania, Ukraine Hungary, Serbia	3,460 - 3,849	c. 4,000	Unknown, some local coordination and harmonisation efforts ongoing	→ / ↑	No robust data from UKR, includes expert estimates
Central European	Austria, Belgium, Czech Republic Germany, Denmark, Luxembourg, Netherlands, Poland	780-1,030	c. 3,000	Some local coordination and harmonisation efforts ongoing, but double counts e.g. between CZ and DE occur	↑	Estimates for POL are not based on packs & pairs, but rather density estimates
Dinaric-Balkan	Slovenia**, Croatia, Bosnia & Herzegovina, Montenegro, North Macedonia, Albania, Serbia, Kosovo*, Greece, Bulgaria	c. 4,000	c. 4,700	Excluded between SVN / HRV; unknown for the rest of the range	↑	GRC only preliminary results; for ALB, XKX*, MKD no estimates available

Iberian	Spain, Portugal	No recent update, but 2007 estimate was 2,500	c. 2,400	Excluded at sub-nations scale (between regions). Likely of minor importance along with Portugal given the large number of packs.	→	For Spain, information on pack numbers will be updated by the end of 2024
Italian peninsula	Italy	1,100 - 2,400	2,557	Excluded Harmonized monitoring (WAG)	↑	Change in methods, first coordinated range wide census
Karelian	Finland	c. 200	310	52 wolves (7 packs) transboundary with Russia	↑	
Scandinavian	Norway, Sweden	c. 430	520	Excluded	↑	
Total (c.)		**17,000**	**23,000**		↑	

Unknown* = *it is not known what number or % of animals are counted in more than one country,* ***Excluded* *= coordinated monitoring excluded double counting; **The estimated number of wolves in transboundary packs is divided by 2 to avoid double counting.*

Table 24: Population estimates for the wolf in Europe by country and population. Note: Country level population estimates may include double counting of transboundary individuals. Where population level estimates were available these were used for the sum here other and in Table 23, several estimates were rounded to reflect uncertainty. For references see Appendix 6.

Wolf population	Year(s)	Estimate	Uncertainty	Details	% of range monitored	Trend	Trend quality	Reference
Alpine and neighboring areas		**c. 2,000**						
Austria	2023	58	minimum, includes 1 pack	mainly dispersers; cummulative per year; includes 12 culled individuals	100	Increasing	Real	Rauer & Selimovic 2024 https://baer-wolf-luchs.at/wp-content/uploads/2024/08/OeZ-Statusbericht-Wolf-2023-1.pdf; https://www.kora.ch/en/news/longest-known-dispersal-of-wolf-in-europe--551
France	2023	1,104	95%CI: 1,000-1,210	almost entire French wolf population	100	Increasing	Real	https://www.consultations-publiques.developpement-durable.gouv.fr/projet-de-plan-national-d-actions-2024-2029-sur-le-a2940.html?lang=fr
Germany	2022/23		only sporadic occurence		NA	NA	NA	DBBW 2024a,b
Italy - Alpine Regions (entire northern Italian wolf population)	2020-2021	952	135 packs	95%CrI: 816–1,120 (112-165 for packs)	100	Increasing	Real	Marucco et al. 2023
Switzerland	2023/24	312	without those killed in 2023/24	genetics	100	Increasing	Real	https://www.kora.ch/en/species/wolf/abundance
Baltic		**c. 3,000**						
Estonia	2023	330	33 packs	minimum number	100	Increasing	Real	Veeroja et al. 2023
Latvia	2023	650	CI for trend: 500-800		100	Increasing	Real	Šuba et al. 2021, Žunna et al. 2023
Lithuania	2023	736	92 packs, Minimum number		100	Increasing	Unknown: maybe more data & real	Špinkytė-Bačkaitienė 2023
Poland	2023	1,369	range: 1,111-1,667		100	Increasing	Real	https://www.gov.pl/web/gios/poiis---monitoring-wilka-i-rysia
Carpathian		**c. 4,000**						
Czech Republic	2022/23	28	6 packs	territories	95	Increasing	Real	no publications
Hungary	2022	50	40-60	expert estimate	100	Fluctating	Real	Monitoring reports from National Parks
Poland	2023	493	range: 406-580		100	Increasing	Real	https://www.gov.pl/web/gios/poiis---monitoring-wilka-i-rysia
Romania	2018	2,750	range: 2,500 - 3,000	minimum number	70	No obvious change	Real	Romania 2020 - Habitat Directive Report
Serbia	2023	10	range: 8-12	expert estimate	60	Fluctating	Real	
Slovakia	2023	700	Range: 668 - 722; Extrapolation from		100	Increasing	Real	R. Rigg unpubl. data; Appendix 3
Ukraine - Carpathians	2019	???	Uncorrected counts: 563	Population estimates are currently based on counts from protected areas, hunting grounds, and forest units, which are not corrected for double counts.				Cherepanyn et al. 2023
Central European		**c. 3,000**						
Austria	2023	31	minimum, includes 5 packs	mainly disperses; 2 transboundary packs with CZ; high turn-over in	100	Increasing	Real	Selimovic & Rauer 2023; https://baer-wolf-luchs.at/verbreitungskarten/wolf-verbreitung
Belgium	2023	20	4 packs		100	Increasing	Real	http://biodiversite.wallonie.be/fr/les-loups-wallonie.html?IDC=6456; Flanders: https://www.vlaanderen.be/inbo/media/2368/wolfmonitoring-22-23.pdf
Czech Republic	2022/23	172	34 packs	minimum number, incuds transboundary packs	70	Increasing	Real	No publication for 2023
Denmark	2024	40	range: 32-42	genetics	100	Increasing	Real	https://www.ulveatlas.dk/nyheder/marts-2024-maanedlig-status-fra-ulveovervaagningen/
Germany	2022/23	1,339	184 packs & 47 pairs	minimum number of all confirmed individuals within territories (439-565 adults individuals)	100	Increasing	Real	DBBW 2024
Luxembourg	2023		sporadic 0-1	minimum number, bycatch of		NA	NA	Schley et al. 2021
Netherlands	2023	50	45-55 (9 packs, 1 lone wolf, few dispersers)		95	Increasing	Real	https://publicaties.bij12.nl/voortgangsrapportage-wolf-13-mei-2024/overzicht-verspreiding-wolf Jasman et al. 2021
Poland	2023	1,686	range: 1,349-2,023		100	Increasing	Real	https://www.gov.pl/web/gios/poiis---monitoring-wilka-i-rysia

Table 24 continues the next page

Table 24 continuation

Wolf population	Year(s)	Estimate	Uncertainty	Details	% of range monitored	Trend	Trend quality	Reference
Dinaric-Balkan		**c. 4,700**						
Albania	2023	???	no robust population estimate available;					
Bosnia and Herzegovina	2017-2023	350	SD: 300-400		95	Decreasing	Real	Boitani et a. 2022
Bulgaria	2019	1,000	Range: 800-1,200	lack of regular monitoring	Unknown	Unknown	NA	Bulgaria 2019 - Habitat Directive Report
Croatia	2018-2019	163	minimum number	genetics	100	No obvious change	Real	Kusak et al. 2019, Kusak et al. 2023
Greece	2023	2,075	2,075 individuals, 255 packs; minimum		16	Increasing	Change in monit	Iliopoulos et al. 2021a,b,c, & 2022
Kosovo*	2023	???	no population estimates available			NA	NA	
Montenegro	2023	???	no population estimates available			NA	NA	
North Macedonia	2024	315	range: 270-360	based on Relative Abundance Index in Marovo NP	25	Unknown		Gonev 2022 - MES report
Serbia	2023	700	range: 800-900	expert estimate	60	No obvious change	Real	
Slovenia	2022/23	117	95%CI: 107-125	corrected for transboundary packs	100	No obvious change	Real	Bartol et al. 2023
Turkey	2023	???	no recent, robust population estimates		<10	NA	NA	Ambarli et al. 2016
Iberian		**>2,400**						
Portugal	2019-2021	295	190 to 400 wolves (based on 60 identified		variable	Decreasing		Pimenta et al. 2023, Nakamura et al. 2021
Spain	2012-2014	2,100	297 packs in 2014; more than 300 packs in 2023		100	Stable / Slight increase	Real	Ministry for Ecological Transition 2022; Ministerio de Agricultura Alimenacion y Medio Ambiente 2014; Update of wolf packs ongoing
Italian Peninsula		**2,557**						
Italy	2021	2,557	95%CI: 2,127-2,844	first representative sampling	35	Increasing	Real	Gervasi et al. 2024
Karelian		**310**						
Finland	2023	310	probability interval: 291-331 individuals or 42 (40-46) packs		100	Increasing	Real	Heikkinen et al. 2023
Scandinavian		**517**						
Norway	2023	67	range: 66-68	genetics, number of reproductions	100	No obvious change	Real	Svensson et al. 2023
Sweden	2023	450	95%CI: 360 - 580		100	Increasing	Real	
Total (c.)		**23,000**						

3.2.4. Wolverine

Main methods to estimate wolverine populations in Europe

The main methods to obtain population estimates for wolverine are heavily dependent on snow-tracking and non-invasive genetic analysis, with camera traps providing additional information in Sweden & Norway, and observations of young of the year in Finland (Table 25).

Table 25: Main monitoring methods for wolverine in Europe. Min = minimum number, Repro = reproduction, CMR = capture-mark-recapture, Pres = presence, YoY = Young of the Year, Obs = observation

Country/Region	Camera traps			Snow tracking				Non-invasive genetics		Scats	Howling surveys		YoY obs	Hunter data		Expert estimate	Other
	Min	Repro	CMR	Min	Repro	Natal dens	Track index	Min	CMR	Repro	Repro	Pres		Obs index	Harvest data		
	Wolverine																
Finland							>75%						<10%				
Norway		25-50%				>75%			50-75%								
Sweden		25-50%				>75%			50-75%								

Most recent population estimates for the wolverine in Europe

The wolverine population in Europe has stayed stable, with a slight increase in the Karelian population, but no obvious trend in the Scandinavian population (Table 26 & 27).

Table 26: Population trend of wolverines in Europe since the last update in 2016.

Population	Countries	Estimate 2012-2016	Estimate 2017-2023	Trend	Comment
Karelian	Finland	200 - 250	400	↑	increasing
Scandinavian	Finland, Norway, Sweden	800 - 1,000	900	→	stable
Total (c.)		**1,000 - 1,250**	**1,300**	↑	

Table 27: Population estimates for the wolverine in Europe by country and population.

Population	Year(s)	Estimate	Uncertainty	Details	% range	Trend	Trend quality	Reference
Karelian		**400**						
Finland	2023	400	390-504 Bayesian probability interval for national estimate	track density on winter transects	100	increasing	Real	Kojola et al. 2023
Scandinavian		**900**						
Finland	2023	50	part of national level estimate	track density on winter transects	50	No obvious change	Real	Kojola et al. 2023
Norway	2023	350	reproductions / natal dens (64)	winter checking of female reproductive areas	100	No obvious change	Real	Höglund et al. 2023
Sweden	2023	500	reproductions (91)		100	No obvious change	Real	
Total		**1,300**						

3.2.5. Golden jackal

Main methods to estimate golden jackal populations in Europe

The monitoring of golden jackals is largely based on passive, unstructured monitoring (e.g., hunting bags or incidental camera trapping) and fragmented, irregular active surveys (e.g., howling surveys or non-invasive genetic analysis of scat samples and saliva from livestock attacks) which prevents a rigorous estimation of population sizes. Combining hunting bags with expert-based assessments, the continental population may now number more than 150,000 individuals, with densities in the core range in the order of one territorial group per 10 km^2 and locally reaching up to five territorial groups per 10 km^2 (Salek et al. 2014).

Most recent population estimates for the golden jackal in Europe

Given the aforementioned limitations and the fact that, once established and breeding, golden jackals are hard to reduce or eradicate, it is more the expansion of the population ranges that is currently monitored rather than population numbers or densities. However, population status can be ranked based on the level of the species establishment.

In the 29 range countries in Europe, the jackal is widespread and breeding in 11 and thought to be widespread and breeding in another 3, localised with local breeding in 3, has few individuals in 5, and has the first individuals recorded in 4 countries (Table 27).

Table 27: Ranking assessment of golden jackal status by country and status.

Country	Population status
Albania	Widespread and breeding
Bosnia & Herzegovina	Widespread and breeding
Bulgaria	Widespread and breeding
Croatia	Widespread and breeding
Greece	Widespread and breeding
Hungary	Widespread and breeding
North Macedonia	Widespread and breeding
Romania	Widespread and breeding
Serbia	Widespread and breeding
Slovakia	Widespread and breeding
Slovenia	Widespread and breeding
Kosovo*	Thought to be widespread and breeding
Montenegro	Thought to be widespread and breeding
Turkey	Thought to be widespread and breeding
Austria	Localised, locally breeding
Estonia	Localised, locally breeding
Italy	Localised, locally breeding
Czech Republic	Few individuals, breeding reported
Germany	Few individuals, breeding reported
Latvia	Few individuals, breeding reported
Poland	Few individuals, breeding reported
Denmark	Few individuals
France	Few individuals
Ukraine - Carpathians	Few individuals (in the plain part of the Zakarpattia region)
Lithuania	Few individuals
Switzerland	Few individuals
Finland	First individuals
Norway	First individuals
Spain	First individuals
The Netherlands	First individuals

4. Literature

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Country	Acknowledgements
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Denmark	www.ulveatlas.dk
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Appendix 3 – Selected examples on details concerning mapping and population estimates

These examples are just meant to illustrate the complexity of the mapping and monitoring methods, which can never be summarised in a simple overview.

Mapping

Example 1: Italy – Peninsula – wolves (Francesca Marucco)

Cut-off value for modelled probability presence

The distribution map for the Italian peninsular is based on an integrated spatial model, based on the data collected during a 7-month sampling campaign in 2020–2021 and the method is described in detail in Aragno et al. 2023 and Gervasi et al. 2024. The distribution map is based on occupancy probabilities, but for this report had to be converted into a presence / absence map.

To do so, the authors considered that the psi (average occupancy estimate per cell) estimated by the occupancy model described in Gervasi et al. 2024 has a range that goes from 0.99 – 0.04, related to an averaged CV per cell. We plotted those values and identified a point where the slope of the curve changes dramatically, with the accuracy decreasing very quickly as psi decreases. That point, which graphically is at psi = 0.2 and CV = 0.85, separates these two groups of cells. Therefore, we applied this cutoff, thereby excluding all cells with psi < 0.2 and CV > 0.85, as cells that do not indicate a sufficient probability of wolf presence.

Example 2: Switzerland – wolf distribution (Ines Morena)

Small corrections sporadic versus permanent

Permanent: presence confirmed in ≥3 years in the last 5 to 7 years or reproduction confirmed at least once within the last 3 years.

Sporadic: (highly fluctuating presence) presence confirmed in <3 years in the last 5 to 7 years.

In some specific areas where there were isolated cells classified as "permanent," we checked whether the evidence of presence concerned isolated wolves or established isolated wolves (living in an area for ≥6 months). We changed those isolated cells to "sporadic" where there was never an established isolated wolf.

Example 3: Slovenia – lynx mapping (Miha Krofel)

Multiple methods

Main method is systematic camera trapping conducted throughout entire area with reproduction and about 90% of the species range in the country. Details on camera trapping (grid size etc. available in Fležar et al. 2023a, 2024). In addition, we are conducting snow tracking and non-invasive genetic sampling, focused primarily in areas with suspected reproduction of translocated animals. We are also conducting questionnaires sent to all hunting clubs in the potential species range, which guides us in where to conduct systematic camera-trapping.

Population estimates

Example1: Greece – wolves (Yorgos Iliopoulos, CALISTO)

Population estimates based on sampling areas and multiple signs

Monitoring was systematic in 48 sampling areas (10.000 km^2) and focused on counting wolf reproductive packs during late summer after births of the year's puppies.

Sampling areas were scattered across the country in a stratified manner to include all habitat types and altitudinal zones.

Multiple methods were used at each site: habitat modelling sign surveys, camera trapping and howling sessions.

Detection probability of wolf reproduction was high after combination of those methods.

Average size of wolf territories was estimated per sample area (dividing each study area surface with number of verified wolf packs) and then used for estimation of the total number of wolf packs in the country after extrapolations.

Average size of reproductive wolf packs was estimated from camera traps including pups of the year (summing maximum number of adults and maximum number of pups located per pack).

Then, the estimated average number of packs multiplied with the average pack size was used to estimate wolf population size in number of individuals.

Potential wolf distribution for the period 2017-2023 was estimated with the use of field data and spatial distribution of wolf livestock depredation events provided by HFIO to 67.000 km^2. Actual wolf distribution for 2023 (occupied area for reproductive wolf packs) is assumed to be 75% of the potential wolf distribution (i.e. 50.000 km^2).

Estimations provided **should be considered preliminary** and may differ from final estimations after analyses (occupancy analysis and habitat modelling) and the inclusion of more field data in 2024.

Example 2: Slovakia – wolves & lynx (Robin Rigg, Slovak Wildlife Society)

Extrapolating local densities to obtain national estimates

The official report with hunters' statistics for the 2023 season is available online (http://www.forestportal.sk/wp-content/uploads/2024/06/Polovnicka-statisticka-rocenka-SR-2023.pdf). It includes hunting ground-based estimates of LC distribution areas which can be used to calculate national estimates based on the densities in a reference area in Liptov. To do so, we used three different extrapolation approaches.

Wolves

Method 1: extrapolation from density in model area

We (Slovak Wildlife Society in collaboration with colleagues at the University of Ljubljana Biotechnical Faculty) conduct annual non-invasive genetic monitoring in a model area (Liptov). We have CMR estimates of wolf numbers in this area for every year except 2021. I used these estimates and the total area of hunting grounds (PR) to calculate an estimate of wolf density in Liptov. I then used these density estimates, and the total wolf distribution area as reported by hunters to estimate the total number of wolves in Slovakia by year. For 2023 the result is 722 (Table 1).

Table 1: Extrapolation of wolf densities from study area to national level – method 1.

Year	Wolf range Liptov			Wolf range Slovakia	
	Wolf numbers Genetic CMR	Size (km2)	Wolf density / 100 km2	Size (Km^2)	National wolf numbers
2017	57	1,903	3.00	18,326	549
2018	53	1,878	2.82	17,807	503
2019	34	1,933	1.76	18,673	328
2020	58	1,933	3.00	19,155	575
2021	NA	1,973	NA	19,395	NA
2022	48	2,007	2.39	19,421	464
2023	72	2,007	3.59	20,136	**722**

Method 2: recalibration of hunter-based estimates for each year separately

Secondly, I compared our genetic CMR estimates with hunters' estimates for the same area ("JKS Liptov" and then calculated the difference as a coefficient ("JKS/CMR") which I then used to recalibrate hunters' estimates for the whole country ("JKS SR"). The result for 2023 is 674 (Table 2).

Table 2: Extrapolation of wolf numbers from study area to national level – method 2.

Year	Wolf range Liptov study area			Wolf range Slovakia	
	Wolf numbers Genetic CMR	JKS Liptov count	Coefficient (JKS/CMR)	JKS SR count	JKS SR / coefficient
2017	57	293	5.14	2621	510
2018	53	284	5.36	2561	478
2019	34	249	7.32	2786	380
2020	58	295	5.09	3099	609
2021	NA	354	NA	3291	NA
2022	48	370	7.71	3606	468
2023	72	436	6.06	4082	**674**

Method 3: recalibration of hunter-based estimates using mean coefficient +/- SE

Finally, instead of calculating a coefficient for each year separately, as above, I calculated a mean coefficient for 2017-2023 (= 6.11 SE +/- 0.47) and used this to recalibrate hunters' estimates for the whole country (Table 3). The result for 2023 is 668 (SE: 620 - 723).

So, my conclusion is that a reasonable estimate for 2023 based on available data is approx. 700 individuals (668 - 722 being the range of estimates obtained using the 3 methods described).

Table 3: Extrapolation of wolf numbers from study area to national level – method 3.

Year	JKS SR	JKS SR / mean coefficient	(-SE)	(+SE)
2017	2621	429	398	464
2018	2561	419	389	454
2019	2786	456	423	494
2020	3099	507	471	549
2021	3291	538	500	583
2022	3606	590	548	639
2023	4082	668	620	**723**

Lynx

I went through available sources of lynx density estimates to recalculate estimates of lynx numbers in Slovakia. I found camera trap-based SCR estimates from 7 different study areas in Slovakia, with a mean density estimate of 1.11 (SE 0.13) independent lynx per 100 km^2 of suitable habitat (Table 4).

Table 4: Lynx densities in 7 study sites in Slovakia.

Year	Study area	Lynx density /100 km2	SE
2016-2017	Muránska planina	1.47	
2017-2020	mean Beskydy	0.55	
2017-2020	mean Javorníky	0.84	
2017-2020	mean Kysuce	1.53	
2017-2018?	Strážovské vrchy	0.97	
2018-2019	Veporské vrchy	1.2	
2019-2020	Vtáčnik	1.18	
	Mean	**1.11**	**0.13**

Using the hunting ground-based map for the total distribution area, I calculated density-based estimates of numbers by year (Table 5).

Table 5: Lynx numbers based on average lynx numbers from CMR estimates.

Year	Lynx distribution area Slovakia (km^2)	N lynx in Slovakia (Distribution area x average density)	SE
2017	21,070	233	28
2018	20,109	222	26
2019	20,365	225	27
2020	21,023	232	28
2021	21,051	233	28
2022	20,785	230	27

I also used a second method: comparing SCR estimates with hunters' estimates in study areas and recalibrating hunters' estimates on the national level. For this, I found 5 different study areas for which the hunter error was mentioned (hunters' estimate / SCR estimate; Table 6).

Table 6: Correction factor for hunter estimates based on CMR estimates.

Year	Study area	Correction factor CF (hunters' estimate / SCR estimate)
2013-2014	Štiavnicke vrchy	5.6
2014-2015	Veľká Fatra	6.9
2016-2017	Muránska planina	6.3
2018-2019	Veporské vrchy	11.7
2019-2020	Vtáčnik	8.3
	Mean	**7.8**

Recalibrating the national hunter estimate using the mean correction factor (CF) of 7.8 (min 5.6, max 11.7), I obtained annual estimates for numbers of lynx in Slovakia (Table 7), which are similar to those obtained using the density-based method.

Table 7: Lynx estimates for Slovakia based on hunter observations and the CMR correction factor.

Year	Hunter estimate	Hunter estimate / mean CF	Hunt. est. / max CF	Hunt. est. / min CF
2017	1,768	228	151	316
2018	1,649	213	141	294
2019	1,688	218	144	301
2020	1,723	222	147	308
2021	1,804	232	154	322
2022	1,743	225	149	311
2023	1,819	**234**	155	325

Sources/references for lynx densities:

Kubala J. et al. (2017) Robust monitoring of the Eurasian lynx *Lynx lynx* in the Slovak Carpathians reveals lower numbers than officially reported. Oryx, doi:10.1017/S003060531700076x.

Smolko P. et al. (2018). Lynx monitoring in the Muránska Planina NP, Slovakia and its importance for the national and European management and conservation of the species. Technical report. DIANA – Carpathian Wildlife Research, Banská Bystrica, Slovakia, 30 pp.

Kubala J. et al. (2019). Monitoring of Eurasian Lynx (*Lynx lynx*) in the Vepor Mountains and its importance for the national and European management and species conservation. Technical report.

Kubala J. et al. (2020). Monitoring rysa ostrovida (*Lynx lynx*) v Strážovských vrchoch a jeho význam pre národný a európsky manažment a ochranu druhu. Technická správa. [Monitoring of lynx (Lynx lynx) in Strážov Mountains PLA and its importance for national and European management and species protection. Technical report]. Občianske združenie DIANA – Výskum karpatskej fauny, Banská Bystrica, Slovensko. (in Slovak)

Kubala J. et al. (2020). Eurasian lynx (Lynx lynx) monitoring in the Vtáčnik Mountains and its importance for the national and european management and conservation of the species. Technical report. Technical university in Zvolen, Zvolen, Slovakia, 26 pp.

Duľa M. et al. (2021). Multi-seasonal systematic camera-trapping reveals fluctuating densities and high turnover rates of Carpathian lynx on the western edge of its native range. Scientific Reports 11: 9236.

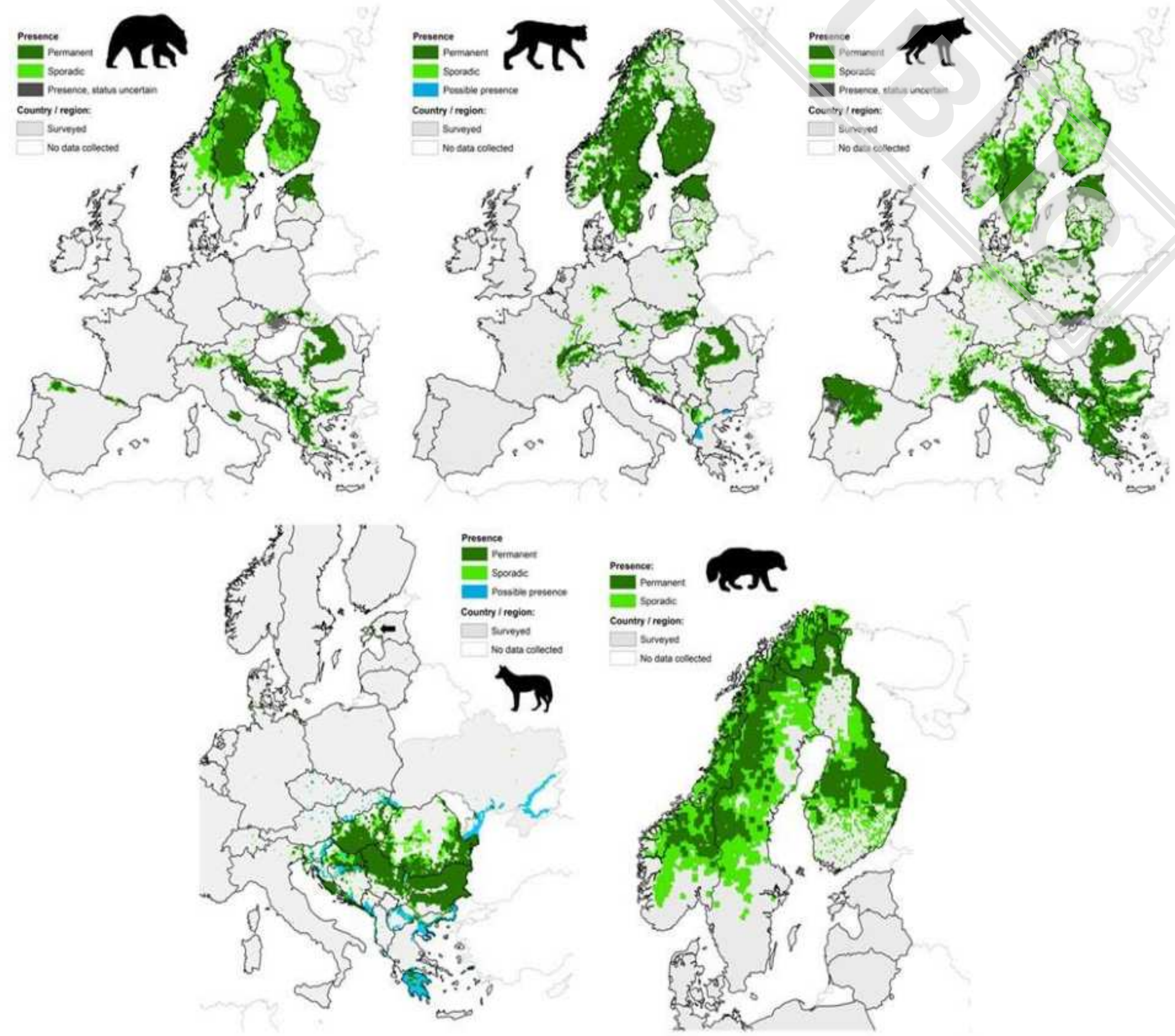
Example 3: Bulgaria – bear (Diana Zlatanova, Faculty of Biology, Sofia University)

Bear tracking transects for population trend

This monitoring with measuring footprints for estimating individual animals has many flaws such as subjectivity in the measurement; control for the error margins during measurements or recording the locations of the tracks; the participants in the monitoring are not always properly trained; etc. Additionally, the climatic conditions in the last few years (dry autumn, when the monitoring is conducted) contribute to a very low track encounter rate introducing additional errors in the trend assessment.

Appendix 4 – Online Questionnaire Mapping (simplified)

LCIE – Update of distribution maps for Eurasian lynx, wolf, brown bear, and wolverine in Europe



BACKGROUND:

For the EU Commission contract N° 09.0201/2023/907799/SER/ENV.D.3 “Support for Coexistence with Large Carnivores”, the Large Carnivore Initiative for Europe (LCIE) has been subcontracted, among other tasks, for task “B.4 Update of the distribution maps” which encompasses: Updating the distribution maps (permanent & sporadic) of wolf, brown bear, European lynx, and golden jackal using the latest available data for the **period 2017 – 2022/23** for the whole EU and adjacent non-EU countries on the level of 10x10 km grid cells (ETRS89-LAEA)with a stronger focus on data quality for each 10 x 10 km cell. LCIE will use the opportunity to also update the population numbers and include the wolverine in the update.

What is new, is that this time we will have to produce two maps:

- Map 1 showing large carnivore presence (which cells have carnivore presence)

- Map 2 showing data quality (which of the presence cells are based on confirmed signs and which are based on extrapolation or soft information)

This questionnaire has 20 questions and will take about 20-30 minutes to fill. It is relevant for the mapping part of the LC status update and asks about data types, sources, and mapping details. The information will be summarised in a meta-document which will be provided together with the updated distribution maps.

Please fill one questionnaire for each species for each country (or region, where data sources and mapping methods vary widely).

Data identifying part

Name:
Country:
Email address:
Full affiliation:

1. What species are you reporting on? Please, fill a different form for each species in your country/region

(1) ❍ Eurasian lynx

(2) ❍ Wolf

(3) ❍ Brown bear

(4) ❍ Wolverine

(5) ❍ Golden Jackal

2. What part of the country are you reporting on here? If different people fill in for different parts of the country or monitoring varies fundamentally between different regions, please fill in a separate form for each area.

(1) ❍ Known species range of the entire country

(2) ❍ Other: _____

Spatial and temporal scale of range monitoring

3. What time period does your species presence map cover?

(1) ❍ 2017 – 2022/23

(2) ❍ Other: _____

4. When does the monitoring period for the species start and end?

_____

5. At what time intervals is species presence monitored?

(1) ❍ Annually

(2) ❍ Cumulative over a regular time period

(3) ❍ Irregular / opportunistic

(4) ❍ Other: _____

6. At what spatial extent is the species range monitored?

(1) ❍ Entire known range

(2) ❍ Entire known range but via rotating monitoring areas

(3) ❍ Only in certain reference areas

(4) ❍ Other: _____

7. Approximately what % of the known species range is monitored by actively looking for species presence to confirm the range:

_____

8. Approximately what % of the known species range is monitored by opportunistic monitoring:

_____

INFO ONLY - Data base for maps - recap on "presence" categories for your information

We will use the following presence categories, which are derived from the SCALP criteria for lynx in the Alps (Molinari-Jobin et al. 2012) but supplement them with two additional data quality information:

1. Confirmed presence signs

- **Category 1 (C1)**: "Hard facts", verified and unchallenged large carnivore presence signs (e.g. dead animals, DNA, verified camera trap images);
- **Category 2 (C2)**: "Confirmed signs", large carnivore presence signs controlled and confirmed by a large carnivore expert (e.g. trained member of the network), which requires documentation of large carnivore signs; and

2. Extrapolated confirmed presence signs

- **Category "buffered":** Confirmed presence signs with a buffer around them based on well documented/ published methods.
- **Category "modelled": confirmed presence signs and modelling** based on habitat suitability and/or proximity criteria based on well documented/published methods. For areas of poor coverage and infrequent monitoring, we will also include:

3. Unconfirmed presence signs - ideally this category is only included where monitoring is extremely fragmented

- **Category 3 (C3)**: Unconfirmed category 2 large carnivore presence signs and all presence signs such as sightings and calls which, if not additionally documented, cannot be verified.
- **Category "Soft":** Extrapolation of large carnivore presence based on interviews questionnaires, and media coverage from 2017-2022/23
- **Category "Past presence"**: Documented presence from the past (but no older than from 2010) and no indication that the situation has changed.

9. What documented signs of the species do you accept as confirmed presence? (basically, SCALP criteria 1 & 2 or an equivalent)

____
____
____
____
____
____

10. On what type of spatial data is your presence data based on?

(1) ❍ Presence is based on the location of confirmed presence signs which were overlaid with the 10 x 10 grid

(2) ❍ Presence is based on buffered confirmed presence signs which were overlaid with the 10 x 10 grid

Please describe buffer size and reason/reference for the size:

(3) ❍ Presence is based on confirmed presence signs and modelling based on habitat suitability and/or proximity criteria which were overlaid with the 10 x 10 grid

Please provide model details and reference:

(4) ❍ Presence is based on larger areas (e.g. hunting grounds) with confirmed presence signs (e.g. hunted individuals) which were overlaid with the 10 x 10 grid

Did you use a cut-off value for intersection (e.g, > 10% overlap with 10x10 cell):

(5) ❍ Presence in part of the range is based on unconfirmed presence signs, or assumed presence based on interviews, questionnaires, and media reports, or documented past presence (this past presence cannot be older than from 2010)

Please provide a short description of the unconfirmed / soft data you used:

INFO ONLY - Presence status - recap on "presence status" categories for your information

We will again aim for presence maps where we can distinguish between two presence levels:

- **Permanent** = suggesting an established population which is reproducing
- **Sporadic** = suggesting only occasional presences of dispersers or lone individuals

Where this distinction is not possible, but presence has been confirmed, we will use

- **Present** = no information about the presence status possible

However, finding a common definition that fits all monitoring circumstances is difficult and the distinction will in parts require expert assessment. Here are the most common scenarios from the last mapping round:

1) For **countries where the known annual species range is monitored annually**, the distinction between permanent and sporadic can be made based on how reliably the species was detected in a cell over the 5–7-year monitoring period:

- **Permanent** = presence confirmed in ≥ 3 years in the last 5 - 7 years OR reproduction confirmed at least once within the last 3 years
- **Sporadic (highly fluctuating presence)** (presence confirmed in <3 years in the last 5 years OR in <50% of the time)

2) For **countries where the probability of species presence is modelled** based on present signs in combination with habitat parameters and distance rules, the distinction between permanent and sporadic can be made based on the probability of presence value. As models used for different populations will vary in their approach, **the cut-off values for permanent, sporadic, and absent need to be defined by the national/population level species experts together with the modeller**.

3) For **countries where the known range is covered cumulative over a 5–7-year period** (period since last update), other criteria need to be used such as: comparison to presence in a cell (or adjacent cells) when the same area was monitored last, or buffers around cells with confirmed reproduction to delineate permanent presence from sporadic presence.

However, where monitoring is too fragmented and infrequent so that and no reasonable distinction between permanent and sporadic can be made, just use the category “present” for cells of confirmed presence.

In general, telemetry data of long-distance dispersers out of the known range and once off documentation of individuals outside the known range should be categorised as sporadic presence.

11. What criteria did you use to distinguish between permanent and sporadic presence?

(1) ❍ Use of re-occurring presence (presence confirmed in ≥ 3 years in the last 5 - 7 years in a cell) and/or reproduction confirmed at least once within the last 3 years

(2) ❍ Use of modelled probability of species presence using a certain cut-off value (see 11b)

(3) ❍ Other criteria were used to distinguish between permanent and sporadic (see 111b)

(4) ❍ No distinction was made, and cells were defined as "present" because monitoring is too fragmented and infrequent for making a reasonable distinction between permanent and sporadic presence

11b. Please describe briefly how the distinction between permanent and sporadic was done:

____

____

Trend of species range

12. Has the method to produce species presence map change since the last reporting for 2012-2016?

(1) ❍ No

(2) ❍ Yes , please explain: ______

13. Are there recent official maps of the species distribution range which vary clearly from your map?

(1) ❍ No

(2) ❍ Yes, please explain why and in which way are they differing:_______

14. How is the trend in the species distribution since 2012-2016?

(1) ❍ Increasing

(2) ❍ No obvious change

(5) ❍ Fluctuating

(3) ❍ Decreasing

(4) ❍ Unknown

15. In your opinion, is the trend real or more likely a result of changed monitoring methods?

(1) ❍ Real

(2) ❍ Different method

(3) ❍ Different environmental conditions

(4) ❍ Other _____

Main monitoring method for the species

16. How much of the species known range is approximately mapped with these method(s)?

	Not relevant	<10%	10-25%	25-50%	50-75%	>75%
Dead animals (hunting, culling, traffic, other)						
Non-invasive genetics						
Camera traps						
GPS tracking						
Active snow tracking						
Howling surveys						
Family group monitoring (natal dens, family groups, female bears with coys)						
Confirmed presence signs such as kills and tracks (SCALP C2 equivalents)						
Damage statistics (consisting of a mix of confirmed and non-confirmed records)						
Unconfirmed presence signs						
Questionnaire surveys & interviews						
Past presence signs based on confirmed signs (but not older than from 2010)						
Other						

16a. Please describe your other method

____
____

Data source and data provider part

17. Which publications best describe range the current monitoring in your country/region? Please, list reference with DOI for published papers and provide link to reports where available. Please use a ";" at the end of each reference.

____
____
____
____
____
____

18. Who needs to be listed as co-author in the report and potential subsequent publication? Please, list with full name and affiliation in the online document at: LINK

19. Who needs to be acknowledged as data provider? Please, list with full name and affiliation.

____
____
____
____
____
____

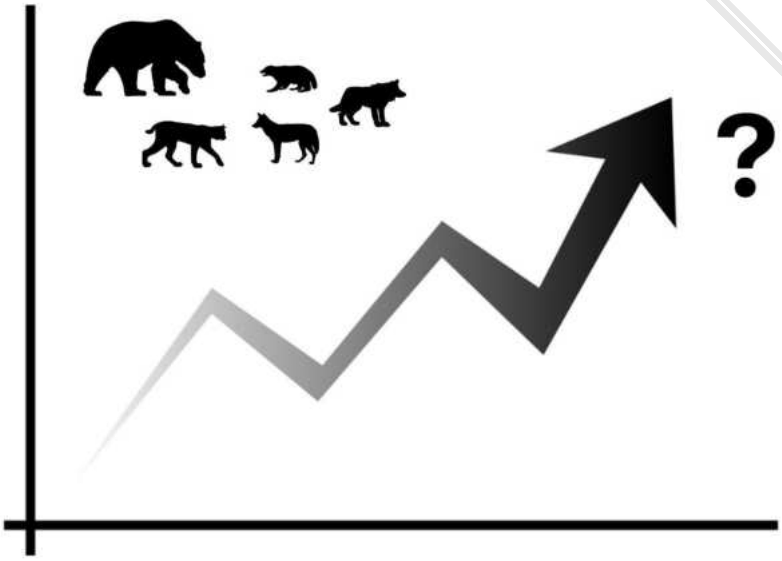
20. Do you have any additional comments:

____
____
____
____
____
____

Thank you!

Appendix 5 – Online Questionnaire Population estimates (simplified)

LCIE – Update on large carnivore status in Europe



BACKGROUND:

For the EU Commission contract N° 09.0201/2023/907799/SER/ENV.D.3 “Support for Coexistance with Large Carnivores”, the Large Carnivore Initiative for Europe (LCIE) has been subcontracted, among other tasks, for task “B.4 Update of the distribution maps”. LCIE will use the opportunity to also update the population numbers.

This questionnaire has 27 questions and will take about 30 minutes to fill. The information will be summarised in report together with the updated distribution maps.

Please fill one questionnaire for each species for each country (or region, where data sources and mapping methods vary widely).

You can abort data entry any time and the data already filled will be saved. You can return to your questionnaire via the individual link you received by e-mail from SurveyXact.

Data identifying part

Name:	_____
Country:	_____
Email address:	_____
Full affiliation:	_____

What species are you reporting on? Please fill a different form for each species in your country/region

(1) ❍ Eurasian lynx

(2) ❍ Wolf

(3) ❍ Brown bear

(4) ❍ Wolverine

(5) ❍ Golden jackal

Population estimate for the species in your country
We will ask for population estimates at national level and where relevant also for population estimates for population segments in your country which belong to different populations as defined by the LCIE.

However, if you only report for a specific region of your country, you can select this option in the first question and will not be asked for national estimates.

1. What part of the country are you reporting on here? If different people fill in for different parts of the country or monitoring varies fundamentally between different regions, please fill in a separate form for each region.

(1) ❍ Entire country - national population

(2) ❍ Specific region (please list name): _____

2a. What is the most recent population estimate for the species in your country/region?

Region	
Year estimated	
Population estimate	
Unit of estimate (e.g., Individuals, packs)	
Measure of uncertainty (e.g., range, 95% CI, SD, SE, minimum number)	
If relevant, conversion factor to number of individuals, else type "No"	
How much of the species known range is approximately monitored (in %)?	____________________________ ______________

2b. How is the current trend of the species population (since 2016)?

(1) ❍ Increasing

(2) ❍ Fluctuating

(3) ❍ No obvious change

(4) ❍ Decreasing

(5) ❍ Unknown (please comment) _____

2c. Do you believe that the population trend since 2016 is real or an artefact of changes in monitoring or management?

(1) ❍ The trend reflects a real population change

(2) ❍ The trend is an artefact of changes in monitoring

(3) ❍ The trend is due to changes in management

(4) ❍ Unknown (please comment) _____

2d. What are the most recent relevant publications / report supporting the species' population estimate(s) in your region? Please list references or links to reports. Please write a ";" after the end of each reference or link.

____

Just for INFO - Sub-national estimates of population segments belonging to different European populations as defined by LCIE –

This section is only relevant if you have more than 1 population in your country. The following main populations have been delineated by LCIE :

- **Eurasian Lynx (11 populations):** Alpine, Balkan, Baltic, Bohemian-Bavarian-Austrian, Carpathian, Dinaric, Harz Mountain, Jura, Karelian, Scandinavian, Vosges-Palatinian
- **Brown bear (10 populations):** Alpine, Baltic, Cantabrian, Carpathian, Central Apennine, Dinaric-Pindos, East Balkan, Karelia, Pyrenean, Scandinavian
- **Wolf (9 populations):** Alpine, Baltic, Carpathian, Central European, Dinaric-Balkan, Italian Peninsula, NW Iberia, Scandinavia, Karelia
- **Wolverine (2 populations):** Karelian, Scandinavian
- **Golden jackal (4 populations):** Adriatic, Continental, Pelopennese, Samos

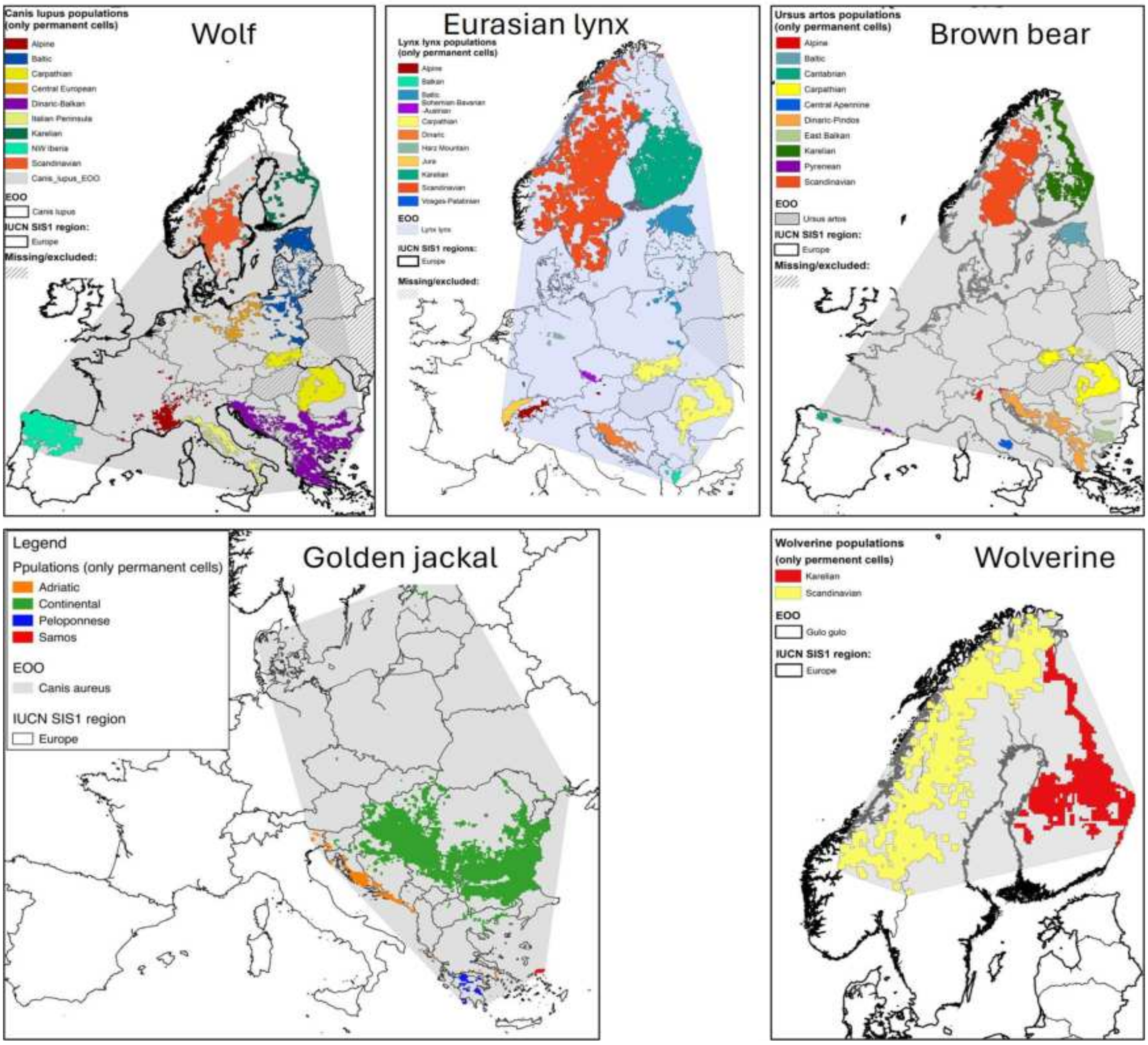


Figure showing the population delineation used for the 2012-2016 update.

4. How many different populations – as defined by LCIE – of the species are found in your country?

(1) ❍ Only 1 population - no need for further details

(2) ❍ 2 populations

(3) ❍ 3 populations

(4) ❍ 4 populations

If available, please provide data on the population size of the different population segments in your country:_________________

……

8. Your population estimates represent what time period of the year? More than 1 answer is possible

(1) ❑ Population estimate in winter (before the next generation is born)

(2) ❑ Population estimate in summer (after the new generation was born)

(3) ❑ Population estmate before the hunting season

(4) ❑ Population estimate after the hunting season

(5) ❑ Information is collected throughout the year

9. Is there any population-level transboundary cooperation in monitoring the species along shared boundaries with neighboring countries? If so, Please briefly describe, else write "no".

____
____
____

10. Do you have an estimates of the number of individuals shared with neighbouring countries (to avoiding double counting cross-border individuals)? If so, Please briefly describe, else write "no".

____
____
____

11. Are there any recent population-level population estimates available, which correct for double counting transboundary individuals? If so, Please briefly describe, else write "no".

____
____
____

Main monitoring method for the species

12. What is the main monitoring method to obtain population estimates and how much of the species range is approximately monitored with this method (% area)?

	<10%	10-25%	25-50%	50-75%	>75%
Camera traps: Minimum number of individuals					
Camera traps: Detection of reproductive units					
Camera traps: Capture-Mark-Recapture estimate of population size					
Snow tracking: Minimum number of individuals					
Snow tracking: Detection of reproductive units					
Snow tracking: Natal den counts					
Snow tracking: Track count index					
Non-invasive genetics: Minimum number of individuals					
Non-invasive genetics: Capture-Mark-Recapture estimate of population size					
Scat surveys: Identify rendezvous sites					
Howling surveys: Confirm reproduction					
Howling surveys: Confirm presence					
Observations: Detection of reproductive units					

(observations of young of the year / family groups)
Observations: Hunter observation index
Hunting bag: population reconstruction based on age / sex structure of harvest
Expert «guestimate»
Other

12a. Please describe your other method

____
____
____

13. Any additional comments on monitoring?

____
____
____

Management

14. Is there an official management plan, action plan, or strategy?

(1) ❍ No

(2) ❍ Yes, please add reference and link _____

15. Is there an official goal for the size of the population?

(1) ❍ No

(2) ❍ Yes

(3) ❍ Don't know

16. What is the population size goal?

_____

17. How is the population size goal interpreted?

(1) ❍ Minimum population size

(2) ❍ Maximum population size

(3) ❍ Not clarified

(4) ❍ Other _____

18. Is there any formal transboundary cooperation in management?

(1) ❍ No

(2) ❍ Yes

19. What is the nature of the transboundary agreement? If possible, provide reference or links.

____
____
____

20. Is the conservation status of the species in your country officially classified as being in favourable conservation status (FCS)?

(1) ❍ Yes

(2) ❍ No

(3) ❍ Not relevant (Non-EU country) / Don't know

21. How has the favourable conservation status defined and what method was used to define it? If possible, provide reference or links.

____
____
____

Data source and data provider part

Please use online spreadsheet: LINK

24. Do you have any additional comments:

____
____
____

Thank you!

Appendix 6 – Most recent publications on population and range estimates

Albania

????, ????. Feasibility study on enhancing connectivity conservation in the PONT Focus Region: Albania and North Macedonia - Description of the selected Connectivity Conservation Areas. Habitat model Balkan lynx and brown bear, Annex to a report.

Catsadorakis, G., Alexandrou, O., Dimidjijevski, D., Hoxha, B., Koutseri, I., Melovski, D., Mertzanis, G., Petridou, M., Shyti, I., Stojanov, A., Trajce, A., 2021. Jackal status in Prespa basin. https://www.researchgate.net/publication/350912059

Bego, F., Trajçe, A., Hoxha, B., 2022. Results on the small, medium and large mammal presence and distribution obtained by sherman and camera-trapping surveys in Gjirokastra region, Albania. PPNEA report. http://dx.doi.org/10.13140/RG.2.2.21207.55202

Hoxha, B., Lama, O., Trajçe, A., 2021. Support to the management of protected areas in border region of Albania, Kosovo, North Macedonia in monitoring of endangered species. PPNEA report. http://dx.doi.org/10.13140/RG.2.2.19529.83045

Hoxha, B., Nezaj, M., Shyti, I., Trajce, A., 2023. Results of the Balkan Lynx Intensive Camera Trapping Survey -Winter/ Spring 2023, conducted in Munella Nature Park and its surrounding areas, Puka-Mirdita Region. PPNEA report. https://www.researchgate.net/publication/380029684

Melovski, D., Shumka, S., Brajanoska, R., Jovanovska, D., Trajçe, A., Nakev, S., Avukatov, V., Hoxha, B., Melovska, N., Custerevska, R., Cveta Trajçe, M.T., Ajola Mesiti Slavcho Hristovski, Pavlov, A., Koci, K., Zhebo, A.S.E., Pandurska-Dramikjanin, F., Mahmutaj, E., Velevski, M., 2022. Feasibility study on enhancing connectivity conservation in the PONT Focus Region: Albania and North Macedonia - Description of the selected Connectivity Conservation Areas. PPNEA report, http://dx.doi.org/10.13140/RG.2.2.31490.08641

PPNEA - Protection and Preservation of Natural Environment in Albania. 2024. Facebook group. https://www.facebook.com/ppnea

Skrbinšek, T., Jelenčič, M., Konec, M., Hočevar, Š., Pazhenkova, E., Gonev, A., 2021. Analysis of noninvasive genetic samples from brown bears (Ursus arctos) from the transboundary Prespa Basin. Report, University of Ljubljana, Biotechnical Faculty & Macedonian Ecological Society.

Trajçe, B.H.A., Shyti, I., Lama, O., 2021. Results of the Balkan Lynx Extensive Camera trapping Survey – Winter/ Spring 2021, Albania. PPNEA report, http://dx.doi.org/10.13140/RG.2.2.31490.08641.

Alps

Hulva, P., Collet, S., Baránková, L., Valentová, K., Šrutová, J., Bauer, H., Gahbauer, M., Mokrý, J., Romportl, D., Smith, A.F., Vorel, A., Zýka, V., Nowak, C., Černá Bolfíková, B., Heurich, M., 2024. Genetic admixture between Central European and Alpine wolf populations. Wildlife Biology.

Marucco, F., Reinhardt, I., Avanzinelli, E., Zimmermann, F., Manz, R., Potočnik, H., Černe, R., Rauer, G., Walter, T., Knauer, F., Chapron, G., Duchamp, C., 2023. Transboundary Monitoring of the Wolf Alpine Population over 21 Years and Seven Countries. Animals 13. 10.3390/ani13223551

Molinari-Jobin, A., Kéry, M., Marboutin, E., Marucco, F., Zimmermann, F., Molinari, P., Frick, H., Fuxjäger, C., Wölfl, S., Bled, F., Breitenmoser-Würsten, C., Kos, I., Wölfl, M., Černe, R., Müller, O., Breitenmoser, U., 2018. Mapping range dynamics from opportunistic data: spatiotemporal modelling of the lynx distribution in the Alps over 21 years. Animal Conservation 21, 168-180.

Molinari-Jobin, A., Urs, B., Breitenmoser, C., Rok, Č., Drouet-Hoguet, N., Fuxjäger, C., Kos, I., Krofel, M., Marucco, F., Molinari, P., Naegler, O., Rauer, G., Sindičić, M., Igor, T., Tijana, T., Woelfl, M., Woelfl, S., Zimmermann, F., 2021. SCALP: Monitoring the Eurasian lynx in the Alps and beyond. CATnews Special Issue 14 Autumn 2021.

SCALP, 2024. Status and Conservation of the Alpine Lynx Population (SCALP) Monitoring reports and maps 2005 -2019. https://www.kora.ch/?action=get_file&id=113&resource_link_id=514

Wolf Alpine Group, 2023. The wolf Alpine population in 2020-2022 over 7 countries; Technical report for LIFE WolfAlps EU project LIFE18 NAT/IT/000972, Action C4; https://www.lifewolfalps.eu/wp-content/uploads/2023/05/C4_WAG_Deliverable_C4_2020_2022.pdf

Wolf Alpine Group, 2022. The integrated monitoring of the wolf alpine population over 6 countries. Technical report for LIFE WolfAlps EU project LIFE18 NAT/IT/000972, Action A5; https://www.lifewolfalps.eu/wp-content/uploads/2022/05/A5_Deliverable_Monitoring-Standards-of-the-Wolf-alpine-population-1.pdf

Austria

3Lynx, 2018. Transnational toolbox for population-level lynx monitoring. Interreg 3Lynx; https://programme2014-20.interreg-central.eu/Content.Node/3Lynx.html

Belotti, E., Engleder, T., Wölfl, S., Mináriková, T., Volfová, J., Bufka, L., Gahbauer, M., Weingarth-Dachs, K., Schwaiger, M., Gerngross, P., Bednářová, H., Strnad, M., Heurich, M., Rodekirchen, A., Wölfl, M., Poledník, L., Zápotočný, Š., Zschille, J., 2023. Lynx Monitoring Report for the Bohemian-Bavarian-Austrian Lynx Population in 2019/2020. Report prepared within the 3Lynx project, funded by Interreg CENTRAL EUROPE programme. https://programme2014-20.interreg-central.eu/Content.Node/D.T2.2.2-Lynx-Monitoring-Report-BBA-LY19-FINAL.pdf

Engleder, T., Fuxjäger, C., 2014. Annual distribution maps Lynx Austria. https://www.luchsfachleute.at/downloads-links

Hatlauf, J., Böcker, F. 2022. Recommendations for the documentation and assessment of golden jackal (*Canis aureus*) records in Europe. *BOKU Reports on Wildlife Research and Wildlife Management* 27: 1-36.

Hatlauf, J. 2022. Der Goldschakal im Lavanttal – Projektbericht 2022. [The golden jackal in the Lavant Valley - Project Report 2022.] Hrsg.: Goldschakalprojekt, Institut für Wildbiologie und Jagdwirtschaft (IWJ), Universität für Bodenkultur Wien. 1-12.

Hatlauf, J. 2024. Statusbericht Goldschakal in Österreich [Update status report golden jackal in Austria](in prep).

Hočevar, L., Fležar, U., Krofel, M., 2020. Overview of good practices in Eurasian lynx monitoring and conservation. INTERREG CE 3Lynx report. University of Ljubljana, Biotechnical Faculty, Ljubljana.

Österreichzentrum Bär Wolf Luchs, 2024. Monitoringstandards für den Wolf in Österreich, Grundlagen und Empfehlungen (Standards for the monitoring of wolves in Austria, principles and recommendations). Österreichzentrum Bär, Wolf, Luchs. Version 2024. https://baer-wolf-luchs.at/

Rau, R., Selimovic, A., 2024. Statusbericht Wolf 2023: Situation des Wolfs in Österreich (Report on the Status of the Wolf in Austria 2023). Österreichzentrum Bär, Wolf, Luchs. https://baer-wolf-luchs.at/wp-content/uploads/2024/08/OeZ-Statusbericht-Wolf-2023-1.pdf

Selimovic, A., Rauer, G., 2023. Statusbericht Wolf 2021 - 2022: Situation des Wolfs in Österreich (Report on the Status of the Wolf in Austria 2021-2022). Österreichzentrum Bär, Wolf, Luchs. https://baer-wolf-luchs.at

Belarus

Belarus, M.o., 2024. Mammals of Belarus, 2017—2024 at 50 x 50 km grid. https://mammals.by/atlas/index

Smith, A. F., Ciuti, S., Shamovich, D., Fenchuk, V., Zimmermann, B., & Heurich, M. 2022. Quiet islands in a world of fear: Wolves seek core zones of protected areas to escape human disturbance. Biological Conservation, 276, 109811. https://doi.org/10.1016/j.biocon.2022.109811

Belgium

Institute for Nature and Forest Research, 2024. Wolf Monitoring Maps for Flanders per monitoring year. https://www.vlaanderen.be/inbo/de-wolf-in-vlaanderen/wolf/

La biodiversité en Wallonie, 2024. Online maps for wolves in Wallonia. http://biodiversite.wallonie.be/fr/les-loups-wallonie.html?IDC=6456

Bosnia and Herzegovina

Trbojević, I., 2017. Distribution of Grey wolf (Canis lupus L., 1758) in Bosnia and Herzegovina. Bulletin of Faculty of Forestry, University of Banja Luka 1.

Trbojević, I., Pašić, J., Brix, M., Stevanović, O., Trbojević, T., 2020. Population status, protection and management of the brown bear (Ursus arctos) in the Republic of Srpska - human dimension (in Serbian with English abstract). Glasnik Šumarskog fakulteta Univerziteta u Banjoj Luci (Bulletin of Faculty of Forestry, University of Banja Luka) 1, 57-74.

Trbojević, I., Penezić, A., Kusak, J., Stevanović, O., Ćirović, D., 2020. Wolf diet and livestock depredation in North Bosnia and Herzegovina. Mammalian Biology 100, 499-504.

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Trbojević, I., Trbojević, T., Malešević, D., Krofel, M. 2018. The golden jackal (Canis aureus) in Bosnia and Herzegovina: density of territorial groups, population trend and distribution range. Mammal Research, 63, 341–348. https://doi.org/10.1007/s13364-018-0365-1

Trbojević, T. 2020. Status zlatnog šakala (C*anis aureus* L., 1758) i potreba za planom upravljanja populacijom u Bosni i Hercegovini. [The status of the golden jackal (Canis aureus L., 1758) and the need for a population management plan in Bosnia and Herzegovina]. Master thesis. Faculty of Ecology, Independent University of Banja Luka. (in Serbian)

Bulgaria

Bulgaria, 2020. Habitat Directive Art. 17 reporting for 2013-2018 https://cdr.eionet.europa.eu/bg/eu/art17/envxhyhkg/

Ministry of Environment and Waters, 2023. National report on the requirements and protection of the environment in the Republic of Bulgaria - 2021. Part of Biological Diversity and National Ecological Network.[In Bulgarian] https://eea.government.bg/bg/soer/2023/5BR.pdf

Spassov, N., Ignatov, A., Mihaylov, T., 2023. New evidence for the recent presence of the lynx, *Lynx lynx* (Linnaeus), in Western Stara Planina Mountains, Bulgaria. Historia naturalis bulgarica 45(3), 53-56.

Serbezov, R., Spassov, N., 2023. Status and Numbers of the Brown Bear (*Ursus arctos L.*) in Bulgaria. Animals (Basel) 13(8), 1412.

Carpathians

Hackländer, K., Frair, J., Ionescu, O., 2021. Large Carnivore Monitoring in the Carpathian Mountains. BOKU-Reports on Wildlife Research & Game Management; http://www.carpathianconvention.org/tl_files/carpathiancon/Downloads/03%20Meetings%20and%20Events/Working%20Groups/Biodiversity/12%20meeting/BOKU%20Report%20LC%20Carnivore%20Monitoring.pdf

Croatia

Gomerčić, T., M. Sindičić, D. De Angelis, I. Topličanec, Kusak, J. , 2023. Procjena parametara potrebnih za ocjenu stanja očuvanosti risa i revizija referentnih vrijednosti (Parameter estimation necessary for the assessment of the state of lynx conservation and the revision of reference values). OPKK projekt „Razvoj sustava praćenja stanja vrsta i stanišnih tipova“ - GRUPA 6: „Izrada i razvoj programa praćenja za velike zvijeri s jačanjem kapaciteta dionika sustava praćenja i izvješćivanja“. Veterinarski fakultet Sveučilišta u Zagreb. https://www.haop.hr/sites/default/files/uploads/dokumenti/2024-05/Plan_upravljanja_risom_s_akcijskim_planom_s_Odlukom.pdf

Huber, Đ., Bišćan, A., Reljić, S., Domazetović, Z., Frković, A., Majnarić, D., Majić-Skrbinšek, A., Sindičić, M., Šprem, N., Modrić, M., Lipošćak, M., Žuglić, T., 2019. Plan gospodarenja smeđim medvjedom (Ursus arctos L.) u Republici Hrvatskoj (Management plan brown bear (Ursus arctos L.) in the Republic of Croatia). Ministarstvo poljoprivrede, Monistarstvo zaštite okoliša i energetike, Zagreb. https://poljoprivreda.gov.hr/UserDocsImages/dokumenti/sume/gospodarenje_divljaci/Plan%20gospodarenja%20medvjedom%202019_final.pdf

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Appendix 7 – Most recent management / action plans

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European Commission

Developing methodology for setting Favourable Reference Values for large carnivores in Europe

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This document has been prepared by the Norwegian Institute for Nature Research for the Istituto di Ecologia Applicata and with the contributions of the IUCN/SSC Large Carnivore Initiative for Europe (chair: Luigi Boitani) and other experts as partial fulfilment of contract N°09.0201/2023/907799/SER/ENV.D.3 from the European Commission.

Developing methodology for setting Favourable Reference Values for large carnivores in Europe

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Contents

Foreword

This report has been developed by the Norwegian Institute for Nature Research with the support of Istituto di Ecologia Applicata and with the contributions of the IUCN/SSC Large Carnivore Initiative for Europe (chair: Luigi Boitani) as well as other experts.

The European Commission issued a a call for tenders (ENV/2023/OP/0019) "Support for Coexistence with Large Carnivores" in 2023. The resulting contract N° 09.0201/2023/907799/SER/ENV.D.3 was awarded to a consortia of Istituto di Ecologia Applicata, Adelphi Consult and Callisto. The Norwegian Institute for Nature Research (NINA) were allocated a subcontract for a specific task (B3 – Assessment of large carnivores' conservation status). The tasks objective were specified as "*Development of a specific, operational methodology to define and quantify the Favourable Reference Values for the species wolf, brown bear, European lynx and golden jackal*".

The report covers three inter-related aspects of the topic, including "Exploring the conceptual basis of setting Favourable Reference Values", developing a "Methodological toolkit for setting Favourable Refence Values" and illustrating the consequences of these approaches via "Scenarios of conservation targets: setting Favourable Reference Values under different decisions". The report was coordinated by NINA, but involves the input of the Istituto di Ecologia Applicata , the IUCN/SSC Large Carnivore Initiative for Europe (chair: Luigi Boitani) as well as other experts within specific fields. Discussions were held in a series of online workshops and by email exchange. Specifically, significant contributions were made by Luigi Boitani, Yorgos Iliopoulos, Juan Carlos Blanco, Ilka Rheinhardt, Joachim Mergeay, Robin Rigg, Djuro Huber, Igor Trbojevic, Alexander Trajce, Jonas Kindberg, Diana Zlatanova, Tomaz Skrbinsek, Astrid Strønen, Nuria Selva, Manfred Wölfl, Arie Trouwborst and Petra Kaczensky. In addition, we would like to thank Katharina Steyer, Robert Ekblom, Nikolas Dussex, Sami Niemi, Scott Mills, Madeleine Nyman, Gunnar Glörsen, Peep Männil, Ilpo Kojola, Janis Ozolins, Pierre-Yves Quenette, Vaidas Balys, Manuela von Arx, Urs Breitenmoser, Peter Sunde, Claudio Groff, Aleksandra Majic, Hugh Jasman, and Miroslav Kutal for providing information and useful discussions on early drafts. The third draft was commented on in detail by various national experts on Habitats Directive reporting as well as stakeholder representatives. We ae grateful for their critical, but constructive, comments and sharp eyes.

Summary

This report aims to develop new guidelines for the setting of Favourable Reference Values (FRVs), which are needed to assess Favourable Conservation Status (FCS), for the specific context of large carnivores (brown bear, Eurasian lynx, wolf, wolverine, golden jackal) in Europe. The work builds on the *Guidelines for Population Level Management of Large Carnivores in Europe* report that was published in 2008, but takes into account new developments in conservation science, new case law, experience with their implementation, and the rapid development of the conservation status of large carnivores.

The need for these guidelines is underlined by the fact that until now relatively few member states have set quantitative values for their FRVs, and there is a massive degree in variation in the scientific basis for those that have. The need for specific guidance on large carnivores stems from both their specific ecology, with wide ranging movements and transboundary populations, and from their complex and often conflictful relationship with humans.

In the report we explore the conceptual basis for setting FRVs. This involves trying to align best-practice and current scientific concepts with the legal / administrative language of the Habitat Directives and associated guidance documents. In recent years conservation science has made important developments in multiple relevant areas, including a shift away from the science of avoiding extinction to a science of planning for species recovery and long term persistence. This involves a focus on building representation of ecological conditions and building resiliency to changing environments, at least in part by ensuring redundancy. It also involves a greater focus on the long term genetics of populations in addition to shorter term demographic aspects. It is also important to recognise that conservation science has made important steps in mapping and understanding the diverse conflicts that are often associated with large carnivore populations in human-modified landscapes.

As a result of this alignment between science and law / policy we developed a number of conceptual recommendations that are important for developing functional FRVs. These include;

- Recognising FRVs as realistic and achievable targets for population recovery that represent the degree of member state contribution which is required for the collective conservation effort.

- Defining FRVs in terms of genetically effective population sizes aligned with the 50:500 heuristic. The 50 and 500 values refer to the effective population sizes required to minimise short term inbreeding and to enable long-term adaptive capability respectively. Effective population size is a genetical concept, where the effective population size is typically between one third and one tenth of the total population size depending on species ecology.

- Recognising that FRVs, and FCS, are not necessarily absolute values. To be achievable they must be scaled to member state preconditions (size, area of habitat, landuse).

- Accordingly we propose a separation between population level FRVs that are pegged on absolute values associated with genetically effective population sizes (often involving transboundary populations), and member state level FRVs that are scaled to their preconditions as long as the contributions of all member states sharing a population sum to a level that satisfies the population level FRV. In other words, FRVs and FCS are both absolute and relative concepts depending on the scale being considered.

- The need for large population sizes (FRP) requires a renewed focus on range (FRR) at national and international levels and ensuring that there are widely dispersed populations with high degrees of connectivity. Mapping and safeguarding this connectivity are important components of FRR.

- We also propose an additional focus on ensuring that range spans all Natura 2000 sites designated for the species, all relevant biogeographic regions, and all relevant ecosystem types. This helps address aspects

related to the ecological functionality of large carnivores which have remained a neglected component of the definition of FCS.

- This approach requires a high degree of coordination in monitoring across borders, and with a strong focus on monitoring both demographic and genetical properties.

- These efforts would be enhanced by transboundary cooperation and the setting of joint management plans, although we also propose post hoc mechanisms for larger scale assessment based on reports submitted by member states.

We integrate these concepts into simple checklists that can guide the setting of the FRVs that are necessary to reach FCS at both population and member state levels. In addition, we provide illustrative scenarios of how current distributions relate to Natura 2000 sites and biogeographic regions, as well as illustrating how different degrees of connectivity and different parameter choices would influence the size of populations required to reach the recommended effective population sizes for the different species.

If these concepts are followed it should secure the long term conservation of large carnivores in Europe. The requirements can be jointly met through transboundary cooperation which shares the effort across member states, and for most populations can be realistically achieved.

Abbreviations used throughout the text

Institutional	
Bern Convention	Convention on the Conservation of European Wildlife and Natural Habitats (19 September 1979)
Birds Directive	Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 on the conservation of wild birds
CJEU	Court of Justice of the European Union
CoE	Council of Europe
EC	European Commission
EEA	European Environment Agency
EU	European Union
GBF	Kunming-Montreal Global Biodiversity Framework adopted in 2022 by the Conference of the Parties (COP15) to the Convention on Biological Diversity
Habitats Directive	Council Directive 92/43/EEC of 21 May 1992 On the Conservation of Natural Habitats and of Wild Fauna and Flora
IUCN	International Union for the Conservation of Nature
LCIE	Large Carnivore Initiative for Europe, an IUCN's SSC SG
MS	Member State of the European Union
Natura 2000	Natura 2000 protected area network
Nature Restoration Law	Regulation (EU) 2024/x of the European Parliament and of the Council on nature restoration and amending Regulation (EU) 2022/869
SSC SG	Species Survival Commission Specialist Group (IUCN)
TFEU	Treaty on Functioning of the European Union
Conceptual	
CV	Current Value (the size of the current population or range)
FCS	Favourable Conservation Status
FRP	Favourable Reference Population
FRR	Favourable Reference Range
FRV	Favourable Reference Value
GSS	Green Status of Species, and IUCN assessment procedure
HWC	Human Wildlife Conflict
MVP	Minimum Viable Population
N_{tot}	Total population size, all age classes, both sexes
N_b	Number of breeding events, breeding units
N_c	Census Population Size (number of mature individuals)
N_e	Effective Population Size
N_i	Number of independent individuals (not following their mother)
PVA	Population Viability Analysis

Introduction

International conservation agreements have been instrumental in halting the declines and fostering the recovery of many species around the world (Trouwborst et al. 2017a). Among the diverse international instruments that exist for biodiversity conservation, the European Union's Habitats Directive (and its sister Birds Directive) stand out because of their strong foundation in super-national law where an international court has the power to impose sanctions for non-compliance on member states. This requires clear definitions of concepts such that compliance can be measurable in objective and repeatable manners. Even when disputes do not enter the legal system, it is highly desirable that legal instruments intended to achieve specific objectives are set-up in a manner that allows assessment of progress towards these goals. Although this is of importance for all species, it is especially important for controversial species whose conservation has large socio-economic impacts on stakeholders and livelihoods because of the need to reconcile biodiversity conservation with other policy agendas and social justice (Milner-Gulland 2024, Zimmermann et al. 2023).

Large carnivores (a collective term for brown bear *Ursus arctos*, wolf *Canis lupus*, Eurasian lynx *Lynx lynx*, wolverine *Gulo gulo* and more recently golden jackal *Canis aureus*) are one such species group. On one hand they are a species group that have responded to improved legislative protection through dramatic recoveries and expansion across the continent. On the other hand their conservation is associated with a diversity of challenges and conflicts where their conservation can have impacts on livelihoods and even human safety. Legal controversies around their conservation have until now mainly focused on cases of derogation from strict protection, but there is an increasing degree of discussion around how far member states have to go in the recovery of the species on their territory. Until now, the accumulated guidance and case law around the means of conservation (i.e. derogation vs protection) has been much clearer than that surrounding the issue of the goals of conservation (centered on the Habitats Directive's concept of Favourable Conservation Status (FCS) and its associated concepts of Favourable Reference Values (FRVs)). Derogation and FCS assessments are interlinked to the extent that one condition of derogations is that they not impede the achievement / maintenance of FCS. Although the Habitats Directive provides a conceptual definition of FCS in Article 1(i) it has proven hard to translate this into measurable parameters and directly equate it with established ecological concepts.

In this report we explore this issue of setting conservation objectives for large carnivores in Europe in three steps. Firstly, we will explore the conceptual basis of Favourable Reference Values and try to connect them to other and more measurable benchmarks that are well established in conservation biology. Secondly, we will propose a set of practical approaches to operationalise these concepts. Thirdly, we will illustrate the consequences of different choices for some real world examples.

Methodology

This report has used a desk-top and virtual workshop approach. Firstly, the literature (both the limited peer-reviewed literature and the wider grey, or technical report, literature) on the specific concepts associated with FCS and FRVs has been reviewed. Gathering this involved taking contact with scientists and administrators in multiple EU countries to gather hard-to-find reports and overviews of current practices. It should be noted that many documents have been automatically translated using online systems (Google Translate and the built in function in Microsoft Word). Secondly, we have reviewed the emerging scientific literature surrounding the topic of setting goals for species population recovery and associated ecological concepts such as population viability and genetical effective population size. Thirdly, we have tried to integrate these ideas into operational proposals that are applicable in a European setting. Fourthly, we have then discussed these in a series of online workshops with members of the Large Carnivore Initiative for Europe (LCIE) and other external experts within specific areas of competence (law and genetics). Fifthly, a third draft of the report was circulated among members of the Habitat Directives' Working Group on Reporting and key stakeholders for comments. A summary of these comments and our responses to them are available in Appendix 2. .

The report has been informed by a previous process that in 2008 produced a set of guidelines for large carnivore population level management (Linnell et al. 2008) which was developed after extensive consultation with stakeholders and competent authorities across Europe. However, we have critically reassessed these in light of the last 16 years of scientific and policy developments. The report has been developed parallel to the updated mapping of large carnivore populations conducted as Task B.4 of the same contract (Kaczensky et al. 2024).

This document is intended to be a presentation of the **best available, policy-relevant, interdisciplinary science**. It is written with the intention of being applied within the existing legal framework of the Habitats Directive. We have endeavoured to develop interpretations of key concepts that are feasible. As such, we have endeavoured to navigate the legal space defined by our reading of the Directive and the accumulating body of case law from the CJEU (Table 1) as well as legal and policy relevant scholarship (e.g. Christiernsson 2019, Darpo 2011, 2020, Epstein 2016, 2017, Epstein & Kantinkoski 2020, Epstein et al. 2016, 2019, Eriksen et al. 2020, Fleurke 2024, Hiedanpää 213, Hiedanpää & Bromley 2013, Köck 2019, Schoukens 2022, Trouwborst 2010, 2014, 2018, Trouwborst et al. 2015, 2017a,b,c) as well as other guidance documents produced by the EC and external contractors on their behalf (e.g. Bijlsma et al. 2019a,b, Van Eldik et al. 2024). The guidelines introduce, discuss and operationalise many concepts, and build on scientific insights, that were not available to the authors of the Habitats Directive when it was drafted and hence were not referred to in the text of the directive. However, we feel that it is appropriate to introduce them as they are the necessary underpinnings of the conservation strategies that are necessary to achieve the intentions which the directive advocates.

It is important to point out that this document is not legislative in character and is not of a binding nature. As such this document reflects only the views of its authors. In accordance with the EU Treaties, it rests with the Member States to choose the form and methods of achieving the objectives of the Habitats Directive. Ultimately **only the CJEU can decide** whether specific policies fall inside, or outside the law. Therefore, the guidance provided will need to evolve in line with any emerging jurisprudence on this subject.

Table 1. Accumulated CJEU case law on large carnivores (based on searches of https://curia.europa.eu/).

Date	Case Number	Location	Species	Topic
Pending	C-27/24	Italy	Bear	Request for preliminary ruling on questions related to derogation from strict protection
AG opinion available, judgement pending	C-629/23	Estonia	Wolf / Bear /Lynx	Request for preliminary ruling on questions related to favourable conservation status and scale of assessment
2024	C-436/22	Spain	Wolf	Request for preliminary ruling on questions related to monitoring, derogation and assessment
2024	C-601/22	Austria	Wolf	Preliminary ruling on questions associated with derogations from strict protection for wolves
2020	C-88/19	Romania	Wolf	Preliminary ruling on questions associated with derogations from strict protection for wolves
2019	C-674/17	Finland (Tapiola case)	Wolf	Preliminary ruling on questions associated with derogations from strict protection for wolves
2011	C-240/09	Slovakia	Bear	Preliminary ruling concerning the rights of environmental NGOs (under the Aarhus convention) to be involved in derogation decisions
2011	C-404/09	Spain	Bear	Ruling on questions related to degradation of a Natura 2000 site designated for bears
2007	C-342/05	Finland	Wolf	Ruling on questions associated with derogations from strict protection for wolves

A: Conceptual background for setting Favourable Reference Values

1 Legal and administrative background for Favourable Reference Values

Biodiversity conservation in Europe has to be seen within the wider context of multiple legal frameworks or laws and conventions that include, but are not limited by the Bern Convention, the Bonn Convention, the European Landscape Convention, CITES, the global Convention on Biological Diversity, the global Biodiversity Framework, the EU Biodiversity Strategy, the Habitats Directive and the recently enacted Nature Restoration Law. Combined, these set a very high level of ambition for European nature conservation that goes far beyond preventing extinction towards mandating large scale ecosystem restoration. Among these, it is the Habitats Directive that has the most direct bearing on setting policy frames for large carnivore conservation within the EU. The ambitious objective of the Habitats Directive is to ensure the long-term conservation of the wild species and habitats of community concern. Specifically Article 2(1) states that "*The aim of this Directive shall be to contribute towards ensuring bio-diversity through the conservation of natural habitats and of wild fauna and flora in the European territory of the Member States to which the Treaty applies*". The Directive formulates this goal through two overarching requirements on the Member States:

- "*Measures taken shall be designed to maintain or restore, at favourable conservation status, natural habitats and species of wild fauna and flora of Community interest*".

- "*Measures taken pursuant to the Directive shall take account of economic, social and cultural requirements and regional and local characteristics*".

The term of **Favourable Conservation Status** (FCS) is defined for species in Article 1(i) of the directive in terms of an overall explanation and three criteria;

- "*Conservation status of a species means the sum of the influences acting on the species concerned that may affect the long-term distribution and abundance of its populations within the territory referred to in Article 2;*

- ***Population dynamics data** on the species concerned indicate that it is maintaining itself on a **long-term basis** as a **viable component** of its natural habitats, and*

- *The **natural range** of the species is neither being reduced nor is likely to be reduced for the foreseeable future, and*

- *There is, and will probably continue to be, a sufficiently large **habitat** to maintain its populations on a long-term basis*."

Member States have the obligation to maintain species at FCS if they are already there, or restore them to this status if they are not yet at that status (Article 2(2)).

The overall **qualitative** objective of the legislation and criteria used to describe it are intuitively easy to understand. However, Favourable Conservation Status is not further defined at any point, in either the directive's text or subsequent case law from the Court of Justice of the European Union (CJEU), in concrete or **quantitative** terms that can be measured or unambiguously benchmarked against established scientific concepts. This has led to degree of discussion among conservationists, administrators and governments concerning whether or not this status has been reached or how far member states (MS) have to push species recovery to satisfy their obligations to the European community. For species on Annex V (Protected – "*species of Community interest whose exploitation may be subject to management measures")* of the directive this concept is crucial as it represents the lower acceptable limit of their conservation ambition and for species on Annex IV (Strictly Protected – "*animal and plant species of community interest in need of strict protection*") it is an important concept when assessing the validity of derogations from strict protection (Darpö 2019, 2020, Epstein et al. 2019). Guidance documents and legal scholarship point out that the Directive's goals are not meant to be minimum values, but rather are meant to be **ambitious and precautionary** with respect to the risk of environmental harm (Mehtala & Vuorisalo 2007).

Member States also have a "**surveillance**" (monitoring) obligation (Article 11) and a reporting obligation (Article 17) where they have to report their progress towards reaching FCS for the species and habitats included on the annexes of the Directive.

In order to introduce a greater degree of structure and consistency in reporting, the European Commission administratively introduced the concept of **Favourable Reference Values** (FRVs) for the 2007-2012 and 2013-2018 reporting cycles (endorsed by Habitats Committee in 2004). The idea is that the FRVs are measurable ways to assess progress towards reaching FCS. Two FRVs were introduced for species, namely **Favourable Reference Population** (FRP) and **Favourable Reference Range** (FRR), to measure population and range characteristics, respectively. The concepts are defined as;

Favourable reference range is the *"range within which all significant ecological variations of the species are included for a given biogeographical region and which is sufficiently large to allow the long-term survival of the species*." (Art 17 explanatory notes).

Favourable reference population is the *"population in a given biogeographical region considered the minimum necessary to ensure the long-term viability of the species*." (Art 17 explanatory notes).

Furthermore, for both values it is stated that the FRVs *"… must be at least the size of the population when the Directive came into force ….*" or "*…must be at least the range (in size and configuration) when the Directive came into force …*". Guidelines also underline that there is an interlinkage between range and population values and that they should be set in an iterative process.

Box 1 quotes full definitions and a set of principles for setting FRVs that were included in the current guidelines for Article 17 reporting (June 2023) from the European Commission to member states.

In current practices, member states are required to define FRVs for each of the species (and habitats) present in their national area and compare these to the current values (CVs) for each species as a way of establishing the extent to which they have reached FCS. Because of this central role in reporting and assessment it is obvious that the way in which FRVs is set can have large consequences.

Box 1 Definitions and General Principles for setting Favourable Reference Values (extracted from the EC document on "Guidelines on concepts and definitions: Article 17 of Directive 92/43/EEC. Reporting period 2019–2024")

Definitions (pages 20-21)
Favourable Reference Range = "*Range within which all significant ecological variations of the habitat/species are included for a given biogeographical region and which is sufficiently large to allow the long-term survival of the habitat/species; favourable reference value must be at least the range (in size and configuration) when the Directive came into force14; if the range was insufficient to support a favourable status the reference for favourable range should take account of that and should be larger (in such a case information on historic distribution may be found useful when defining the favourable reference range); 'best expert judgement' may be used to define it in absence of other data".*
Favourable Reference Population = *"Population in a given biogeographical region considered the minimum necessary to ensure the long-term viability of the species; favourable reference value must be at least the size of the population when the Directive came into force; information on historic distribution/population may be found useful when defining the favourable reference population; 'best expert judgement' may be used to define it in absence of other data."*

Principles (pages 21-22)
"The following general principles should be taken into account in the process of setting FRVs:
• FRVs should be set on the basis of ecological and biological considerations;
• FRVs should be set using the best available knowledge and scientific expertise;
• FRVs should be set taking into account the precautionary principle and include a safety margin for uncertainty;
• FRVs should not, in principle, be lower than the values when the Habitats Directive came into force, as most species have been listed in the Annexes because of their unfavourable status; the distribution (range) and size (population) at the date of entry into force of the Directive does not necessarily equal the FRVs; Or in exceptional cases (for example of species with overpopulations as result of non-conservation artificially feeding or of species which population is increasing since the Directive came into force and which are harmfull to other protected species) the favourable reference population (FRP) should be lower than the current population.
• FRV for population is always bigger than the minimum viable population (MVP) for demographic and genetic viability;
• FRVs are not necessarily equal to 'national targets': 'Establishing favourable reference values must be distinguished from establishing concrete targets: setting targets would mean the translation of such reference values into operational, practical and feasible short-, mid- and long-term targets/milestones. This obviously would not only involve technical questions but be related to resources and other factors';
• FRVs do not automatically correspond to a given 'historical maximum', or a specific historical date; historical information (e.g. a past stable situation before changes occurred due to reversible pressures) should, however, inform judgements on FRVs;
• FRVs do not automatically correspond to the 'potential value' (carrying capacity) which, however, should be used to understand restoration possibilities and constraints"

Factors to be considered (page 21)
• "*Current situation and assessment of deficiencies, i.e. any pressures/problems;*
• Trends (short-term, long-term, historical, i.e. well before the Directive came into force);
• Natural ecological and geographical variation (including genetic variation, inter- and intra-species interactions, variation in conditions in which species occur);
• Ecological potential (potential extent of range, taking into account physical and ecological conditions);
• Natural range, historical distribution and abundances and causes of change, including trends;
• Connectivity and fragmentation.
• Requirements for populations to accommodate natural fluctuations, allow a healthy population structure, and ensure long-term genetic viability;
• Migration routes, dispersal pathways, gene flow, population structure (e.g. continuous, patchy, metapopulation)".

2 Current practices in setting Favourable Reference Values

Each reporting cycle is accompanied by an extensive set of guidelines (see Box 1 for the current cycle). However, despite having existed for two decades, and having been through two complete reporting cycles (with the third currently underway), there is still considerable uncertainty surrounding the setting of FRVs. There has been relatively little formalised and structured research take into these concepts (Bonelli et al. 2021, Brambilla et al. 2011, Green et al. 2020). Green et al. (2020) went as far as to state that “*In our view, there is not yet any adequate, quantitative method to calculate a threshold for favourable population size to contribute to wider assessments of species’ conservation status*”.

Three large technical reports have analysed existing pattens (McConville & Tucker 2015, Van Eldik et al. 2024) and suggested more rigorous approaches (Bijlsma et al. 2019a,b). Currently there are many different approaches in use. Up to the publication of these guidelines in many cases no quantitative values are set, with countries simply choosing to say that the FRVs are “unknown”. In other cases countries used “**operators**”, or symbols such as “>” or “”>>”, “≠” or “=”, to refer to the relative position of FRVs with respect to Current Values (CVs). Only a minority of cases use **numeric values**. When numeric values are set, they come from a huge diversity of different procedures including, (1) expert assessment, (2) various baselines / references, or (3) model based approaches.

Although the use of baselines and models appears on the surface to offer a greater degree of standardisation, there may still be degrees of subjectivity and variation involved. **Baselines** taken at different periods in history can have huge impacts because of variation in the timing of human impacts on different species (Crees et al. 2016). **Models** also involve huge variation in model structure and different parameter settings, and in the suggestion that their outputs be “upscaled” by undefined factors.

A certain degree of **diversity in approaches** is understandable considering that the Habitats Directive concepts and reporting structures are designed to cover thousands of species as different as plants, beetles and bears for which the underlying ecology, human impacts and amount of data available varies dramatically. However, one needs to be mindful of the fact that the diversity of approaches also means that for any given species in any given country it may be possible to generate widely different FRV values depending on which approach is adopted, and which specific parameters are chosen. For example, different historical baselines (with associated estimated reference values) could produce very different FRVs as species have declined and recovered under changing human activities and policies for millennia. Although a lack of harmonisation / standardisation is not automatically a problem for some species and may well fall within member state discretion, it is important in cases where there is a need for extensive transboundary cooperation.

Bijlsma et al. (2019a,b) go a long way to set up a structured work flow for setting FRVs. Involving (1) developing a summary of the species specific ecology and conservation status and a clear narrative of **threat** and **recovery potential**, and (2) following a decision-making tree to arrive at decisions about best approaches to adopt. Their description of the background information needed is extensive and motivates careful consideration and classification of many aspects of species ecology and conservation status, especially the critical issue of adjusting the scale of assessment to the way in which different species use space.

When it comes to setting FRVs, Bijlsma et al. (2019a,b) divide approaches into two broad categories.

Firstly, they discuss the “**reference-based methods**” that are informed by **historical baselines**. Such information is clearly very useful to gain perspective on the present situation (Grace et al. 2019, Nores & Lopez-Bao 2022), but the result can be hugely influenced by the exact period chosen as the baseline and by the decision of how much of this baseline needs to be restored to represent FRVs for population and range?

Furthermore, the question remains as to how to apply this to species like the golden jackal currently expanding beyond their historical ranges (Sanderson 2019), and where it has been argued that there is a clear legal obligation to permit this expansion (Trouwborst et al. 2015). The CJEU has even ruled on this concept, underlining that range is a dynamic concept and even expansions into urban areas may in some cases be considered as natural (see Schoukens 2022 for an example on urban hamsters in Vienna and Case C-88/19 for a case with large carnivores in Romania). Bijlsma et al. (2019a) argue for 50 years before the Directive came into force as a suitable baseline, but for large carnivores this period in the 1940s and 1950s would correspond to their lowest levels ever in many areas (Chapron et al. 2014). Nores & Lopez-Bao (2022) discuss the challenges associated with using historical baselines for wolves in Spain. Finally, historical baselines ignore the reality of ongoing environmental change in the Anthropocene and the fact that nature conservation and natural recovery is leading to novel ecosystems with species appearing in areas for which there is no historical precedent (Corlett 2014).

The only exception where near-historical baselines may have some utility is among some of the newer EU members from **eastern Europe** (in the Balkans, Carpathians and Baltics). Many (but not all species in all countries) of these countries hosted sizeable populations of large carnivores when they entered the EU, and it could be argued following the logic of Bijlsma et al. (2019a) that these represented situations where populations had "**stabilised**" at some level in line with environmental circumstances (ecological and social). In which case these levels could make suitable references for FRVs.

It should also be noted that the status of a species when a country entered the EU (The Directive Value) is often used as a baseline to assess progress in conservation under the auspices of the Habitat Directive, although it is not particularly suited as a baseline in the context of setting FRVs.

Secondly, Bijlsma et al. (2019a) discuss "**model-based approaches**" that can either be based around demographic or **population viability models** or potential range models based around habitat modelling and estimates of carrying capacity. Despite both approaches being well-established scientific methods there is large scope for subjectivity when using them to make concrete goals. Bijlsma et al. (2019a) suggest that the results from population-based models should be **upscaled by a factor** without giving clear guidance on this factor. They suggest a factor of 10 as a rule of thumb, while Green et al. (2020) suggest a factor of 16 for UK birds.

Estimates of **potential habitat** and **carrying capacity** are also fraught with assumptions for species living in human-dominated landscapes, and as for the reference-based models there is always the need to make a subjective cut-off about how much of the potential should be occupied to satisfy FCS. Outstanding challenges relate to situations where carnivores depend heavily on livestock as prey (e.g. semi-domestic reindeer in northern Fennoscandia, or horses, goats and sheep in many parts of southern Europe) or the systems widespread across southeastern Europe where bear populations are subject to supplementary feeding. How should such issues be dealt with when calculating carrying capacity?

Despite their great efforts to bring a systematic approach to the process, the result still unfortunately offers a massive diversity of approaches that can potentially result in widely different assessments. Bijlsma et al. (2019b) present worked examples of their approach, including four large carnivore examples. For wolverines in Sweden they estimate that 600 individuals would be a suitable FRP, for bears in the Apennines they estimate FRP = 250, for wolves on the Iberian peninsula they suggest that FRP > 2500 while for lynx in the Alps they suggest that FRP should just be much greater than the present value (FRP >> 130). There is no clear reason why the different case studies result in such different outcomes. A final challenge is their proposal to adopt a "**sit-and-wait**" outcome when FRVs are based on operators. They formulate the objective as waiting to see when currently naturally expanding populations stabilize, and then adopting this as the FRV. Such approaches are problematic with species on Annex V for example that are subject to hunting because stabilization may happen on levels that humans set, and even for strictly protected species (Annex IV) it offers no guidance for planning recovery, and no measurable scale to assess progress. The only situations where it may be applicable is if there is simply too little knowledge to set more precise values.

Although all approaches may in some way reflect the relative progress of conservation from a less to a more favourable state the problem arises when these concepts become involved in legal proceedings between member states and the CJEU and / or when conservation requirements have large socio-economic impacts. In such cases, the absolute scaling of requirements for FRVs to reach FCS can have significant consequences for individual stakeholder groups and citizens directly affected by large carnivores (e.g. farmers, hunters, the forestry sector, agriculture, transport) and member states.

While different technical approaches will always produce slightly different outputs, the problem is exacerbated by the fact that there is no agreement on the unifying biological concept underlying the concepts of FCS and FRVs across species, or even within species but across different countries.

3 Why are large carnivores a special case?

The special features of large carnivores which make them a challenge for conservation have been dealt with elsewhere in great detail (e.g. Linnell et al. 2008). In this section we shall just list some of the key features relevant for setting FRVs.

- **Massive spatial requirements**. The scale of large carnivore movements make them unique among European terrestrial mammals. Typical home ranges for individual large carnivores, or wolf packs, range from 100 km^2 to several thousand km^2. Because they are often territorial (bears excluded) this results in very low densities. The implication is that their conservation requires very large areas indeed, and that very few European countries, if any, are large enough to host genetically viable populations by themselves. This implies that a transboundary approach to their conservation is essential. Their conservation cannot be achieved in protected areas alone and unavoidably requires that they are allowed to occupy a very large proportion of the landscape of the European continent. These large spatial requirements make their populations highly vulnerable to linear infrastructure (highways, railways, fences) that fragment their ranges.

- **Populations that span borders**. The vast majority of large carnivore populations in Europe span international boundaries requiring cooperation between different countries. In practice this challenge is magnified by the high degree of delegation of management authority from Federal states to their various sub-national entities. While the Habitats Directive provides a certain degree of policy coordination it is still the member states that have the authority to manage and report the segment of the population found within their borders. Many of these populations also embrace the territory of non-EU members, although most of these are now either EU-candidate countries or signatories of the closely related Bern Convention, which provides a high degree of policy coordination. One exception is the Finnish-Russian border, which has shared large carnivore populations under different levels of legal protection.

- **Long dispersal distance**. Young large carnivores are capable of natal-dispersal movements that can exceed 1000 km. Dispersal tendencies vary between species. Wolves have by far the greatest and best documented dispersal potential, with both sexes capable of making movements measured in the 100's of km, up to and exceeding 1000 km. Although less well documented the recent expansion of golden jackals across Europe implies that they too can disperse distances of many 100 km. Individual wolverines, lynx and bears have all been documented making dispersal movements of many 100 km as well, although the average distances are lower than for wolves, and in the case of bears especially there is a clear pattern of females showing far less dispersal than males. The implications for conservation are complex. On one hand the long movements allow spatially disjunct sub-populations to maintain connection even across patches of unsuitable habitat. On the other hand it implies that there is a low predictability in where they will appear, with the possibility of animals turning up in places from which they have been absent for decades. For bears, the low dispersal rates for females implies that populations spread very slowly and subpopulations will remain isolated to a greater degree than for the other species.

- **Broad habitat tolerances.** Large carnivore species in general have broad habitat tolerances. Wolverines occur and breed in 3 biogeographic regions, golden jackals breed in 5 and occur in 7, bears occur and breed in 5, lynx breed and occur in 5 and wolves breed and occur in 6 regions. Within these biogeographic regions they also occur in a wide diversity of habitats, and all show a high degree of tolerance to human habitat modification and human landuse / activity. Wolves again being the most adaptable. The advantage of this broad tolerance is that there is considerable scope for population recovery and restoring connectivity across large areas of the continent. The disadvantage is that it is hard to predict where they will colonise which means they may end up colonising high conflict areas.

- **Well studied**. Compared to many species listed on the directive the large carnivores are exceptionally well studied. Most countries have some form of monitoring in place which enables periodic continent wide assessments of distribution and status (Boitani et al. 2022, Chapron 2014, Kaczensky et al. 2024, Salvatori & Linnell 2005). The species have in general been subject to multiple research projects in different areas of their distribution such that their basic ecology and core parameters are well known. Within this broad picture is a large degree of variation. Golden jackals are less studied than the other four species, and in general populations in southern, and especially southeastern Europe are less studied / monitored than those in the centre and north. Overall, there is a very large pool of knowledge to use to inform conservation planning.

- **Depredation on livestock**. Throughout their distribution area all species are to some extent involved in depredation on livestock, especially sheep, goats and semi-domestic reindeer, but also horses, cattle and domestic dogs (Linnell & Cretois 2018). Bears also destroy beehives. Depredation rates can vary from the anecdotal (e.g. lynx depredation on sheep in the Baltic States) to levels where large numbers of carnivores nutritionally depend on livestock as their main prey (e.g. lynx and wolverine depredation on semi-domestic reindeer in northern Fennoscandia, or wolves in some parts of southern Europe). Depredation on livestock is a major driver of conflict in many areas, and represents a real socio-economic cost for their conservation.

- **Killing of companion and working animals**. Wolf attacks on dogs are well-documented across Europe. In Scandinavia from 1998 to 2017 there were 30.6 attacks on dogs annually, of which 6.8 occurred in Norway and 23.8 in Sweden. The majority (83%) took place during moose and hare hunting. Overall, 90.2% of the attacked dogs were hunting dogs. Most attacks occurred in proximity to wolf territories (72% in Sweden). In the Nordic countries, the use of free-ranging dogs is crucial for achieving wildlife management objectives and targets.

- **Impact on game species**. Wolves prey on wild ungulates, sometimes competing with hunters for the game. In some cases, wolves can have a significant impact on game populations in their range as well as certain hunting modalities. This can lead to challenges for wildlife management as well as significant socio-economic damage.

- **Social conflicts and charisma**. Large carnivores are species that generally have a large cultural role in Europe, they are well known to the public, and people tend to have clear opinions about them. These viewpoints can be very diverse, ranging from extremely positive to extremely negative. Those who are directly affected by large carnivores tend to be more negative than people who are not directly affected by their presence. Wolves and bears are classified as priority species under the Directive. This creates the basis for strong social conflicts concerning the way they should be managed and the extent to which they should be allowed to recover in terms of numbers and distributions. In recent years numerous social science studies have provided clear evidence highlighting that these perceptions and attitudes must be taken seriously and be fully taken account of when devising policy and management options concerning large carnivores, including when discussing higher ambitions and risk associated with the setting of FRVs.

- **A threat to human safety?** There is much debate about the potential danger that wolves pose for human safety. Two extensive reviews have documented their potential risk, but also demonstrated that the risks are very low in modern day European landscapes (Linnell et al. 2002, 2021). Brown bears indisputably represent a potential risk to humans (Bombieri et al. 2019, Penteriani et al. 2017, Støen et al. 2018) and recent years have seen an unfortunate number of high profile episodes in parts of southern and southeastern Europe (Cimpoca & Voiculescu 2022).

Overall this guild of species has shown their enormous potential for recovery across Europe, including in human-modified and human-dominated landscapes (Cimatti et al. 2021, Cretois et al. 2021), however, they are also associated with significant economic, social and cultural costs and high conflicts in some regions that should not be underestimated. Predicting the location of these impacts and conflicts is not easy, and may be fluid over time. Furthermore, all approaches to their conservation require a high degree of transboundary cooperation (between protected areas and surrounding landscapes, between sub-national administrative units, between countries within the EU, between the EU and countries outside the EU) and cross-sectorial policy coordination.

4 Existing practices for setting FRVs for large carnivores

During 2007-2008 the Large Carnivore Initiative for Europe developed a set of guidelines on large carnivores for the European Commission that included a section interpreting FCS and FRVs for this species group. The guidelines were developed by experts, but with extensive consultation from responsible authorities in member states, the Habitats Committee, the European Commission and stakeholders. These guidelines made several clear recommendations;

- Promoting the **population level** as the unit of assessment, which for most populations implied a need for transboundary coordination, on the condition of a binding transboundary agreement being developed.

- **Linking FRP to IUCN Red List criteria** such that the absolute minimum for a FRP would be at a level where the criteria would no longer consider the population to be at risk (i.e. it should be Near Threatened or Least Concern) based on either criteria D (1000 mature individuals) or E (using a PVA to calculate the MVP with <10% chance of extinction in 100 years). The recommendations included the option to accept a higher category of threat if there was adequate connectivity between populations when using criteria D, but not for E.

The **2008 guidelines** were endorsed as best practice by the European Commission as well as being recommended to signatories of the Bern Convention by the secretariat. Since their publication the guidelines have been widely quoted by responsible authorities from member states and there appears to have been a growing acceptance of many of the ideas and principles. However, although they are not a legal requirement it is interesting to note that there are as yet no examples of any countries entering into formal and binding transboundary population level management plans (Blanco 2012, Boitani et al. 2022, Eriksen et al. 2020, Kaczensky et al. 2024), despite the existence of widespread technical cooperation in monitoring and research. One good example of progress on the way is the strategy for the joint German-Czech-Austrian lynx population (Bohemian-Bavarian-Austria population) that was developed by the respective national and local level ministries (Czech Ministry of Environment 2020). A *Framework for Transboundary Cooperation on Management and Conservation of Wolves in Fennoscandia* was signed by the responsible technical agencies of Norway, Sweden and Finland in 2020. The technical agencies of the countries sharing the Alpine wolf population prepared a proposal for a coordinated plan in 2016 (Schnidrig et al. 2016) under the auspices of the Alpine Convention, but it was never adopted by the national governments. The Benelux countries, together with France, Germany and Denmark, are currently initiating a working group to coordinate wolf management in northwest continental Europe. However, despite this encouraging degree of technical engagement there as few politically binding agreements committing countries to a sharing of responsibility.

There has also been a degree of **critique** of the guidelines from scientists and conservationists (e.g. Epstein 2016, Epstein et al. 2016, Laikre et al. 2009). Issues mentioned have included;

- The idea that the IUCN criteria E (<10% extinction risk in 100 years) opens for too high a risk of extinction.

- Not enough attention has been paid to the issue of genetics when discussing viability.

- Not enough attention was paid to ecological functionality.

In other words, these critiques interpret the guidelines as being too focused on avoiding extinction and **not promoting a more ambitious recovery level** in line with directives aims.

These critiques, plus developments in science, policy and on the ground realities imply that it is logical to **revisit the recommendations** from these 2008 guidelines with respect to setting FRVs. Specifically;

- There have been many **advances in conservation science**, concerning the conceptual understanding of recovery (as opposed to avoiding extinction) and within conservation genetics that are relevant for setting FRP.

- The passing of the **EU Nature Restoration Law** in 2024 further enshrines the ambition level for ecosystem, habitat and species restoration.

- The guidelines did not treat ecological aspects of large carnivore recovery in great detail, which lead to a rather narrow focus on **FRR** as being only focused on supporting FRP, and not having additional associated criteria associated with **ecological functionality**.

- The **biogeographical regions** and Natura 2000 sites were not addressed in detail. Recent studies have shown that the sites are relevant for carnivore conservation, but that there is a much greater need to focus on this potential contribution of the sites for carnivores, and of the carnivores to the sites.

- Wolves have recolonised many of the very small and / or heavily **human-dominated countries** in western Europe (e.g. France, Germany, Denmark, the Benelux countries) to a degree that was probably not anticipated when the directive was drafted. This requires a reconsideration of the expected contributions from very small countries to collective conservation objectives and of the way conservation potential is conceptualised in landscapes with high human density and heavily human-modified landscapes.

- The failure of most countries to develop the suggested **transboundary management plans**.

- The increase of border security **fencing** (Linnell et al. 2016, Reljic et al. 2018) and veterinary fencing in response African Swine Fever outbreaks which is increasing fragmentation of habitats, and is dramatically decreasing connectivity between European populations and those further to the east.

Current practices in procedures for setting FRVs in different member states are discussed in Appendix 1.

5 Developments in conservation science

5.1 Towards a science of recovery

The science of ecology and its link to applied topics like sustainable wildlife management and biodiversity conservation has been in constant development during the last century. The tradition of **sustainable management** of wildlife populations is by far the oldest branch of applied ecology relevant for wildlife conservation (Leopold 1937, Redford et al. 2011), historically formed the basis of large carnivore management in most southern and eastern European countries prior to their entry into the EU, and still form the basis of large carnivore populations in areas where they are managed under Annex V and when derogations are issued under Annex IV, as well as their wild ungulate prey across Europe. However, the ideas and experience of wildlife management have become less visible as populations are managed under Strict Protection regimes of Annex IV. In contrast, the structures of the much younger science of Conservation Biology have had greater and more visible influence. The early days of Conservation Biology focused heavily on **avoiding extinction**. This is reflected for example in the IUCN Red List status overviews with their well known categories of Critically Endangered, Endangered, Vulnerable etc. These IUCN criteria were central in the 2008 large carnivore guidelines (Linnell et al. 2008) which aimed to align the legal ideas of FCS and FRVs with avoiding endangerment (i.e. setting FRVs at a level that would not justify classification on a threat level).

Science constantly moves forward, and the last two decades have seen a dramatic shift away from a single focus on extinction avoidance (avoiding an unwanted outcome) to that of **planning for recovery** (articulating a desired outcome). A wide set of scientific papers have discussed questions like “How much is enough? Setting measurable objectives” (Tear et al. 2005), “Moving beyond Population Viability Analysis” (Wolf et al. 2015) and “What does it mean to successfully conserve a species?” (Redford et al. 2011). Redford et al. (2011) identified six properties of a recovered population of a species;

- Self-sustaining demographically and ecologically [i.e. they have access to prey / naturally food] and maintaining critical ecological interactions,
- Genetically robust,
- Have healthy populations,

- Have representative populations distributed across the historical range in ecologically representative settings,
- Have replicated populations within each ecological setting,
- Be resilient across the range – e.g. large metapopulations.

These ideas have been condensed into the heuristic of the “**3 R’s**” (Tear et al. 2005, Wolf et al. 2015);

- **Representation** – present in the full range of ecological settings of a species’ range.

- **Redundancy** – multiple populations in each ecological setting.

- **Resiliency** – ability to persist in the long term in the face of changing threats and changing environmental conditions.

In a move to compliment the well-established Red List for threatened species the IUCN are currently working on a **Green Status of Species** (GSS) assessment procedure to measure the pathway to recovery for species as a result of conservation interventions (Akcakaya et al. 2018, 2019, Grace et al. 2021a,b, Stephenson et al. 2020). In a parallel to the 3 R's, the GSS assessment is based on 3 dimensions of recovery (Akcakaya et al. 2018);

- Viability of the population

- Functionality within the ecosystem

- Representation of different ecological settings

In many ways these new frames represent a return to some of the key concepts of biodiversity which recognise the existence of biodiversity at three levels – genes, species and ecosystems.

These developments within conservation thinking show a strong convergence towards the ideals of the Habitats Directive which has long been cited as representing a forward looking and outcome orientated view of conservation where goals, such as FRPs, have been stated as being much greater than Minimum Viable Populations (MVPs). We explore the links between emerging ecological concepts and the legal concepts of the Habitats Directive in section 1.

5.2 Viability – moving beyond demographics

Most of the conservation biology and applied literature has focused on a critical, but narrow, aspect of the concept of viability. This focus has been on **demographic viability**, which is typically based on a calculation of vital rates (birth rates and mortality rates) to estimate the probability of populations of different sizes becoming extinct over certain time frames. The idea of a minimum viable population is the size of the population that will only have a 5 or 10% chance of becoming extinct over a 100 year time horizon (see Linnell et al. 2008 for a longer discussion of PVAs). Such concepts are central when trying to avoid extinction in a crisis situation with small populations, but say little about planning for long term recovery.

Making the step towards longer term recovery requires focusing much more on the genetic components that underpin a species ability to avoid inbreeding and adapt to environmental change. Instead of focusing on the minimum viable population there is a related concept called **Effective Population Size**, typically represented by the expression N_e. Effective populations size is a formal concept in genetics that is more complex than just the number of individuals that breed as it also takes into account the variation in which different individuals contribute to the next generation and how this influences the genetic structure (heterozygosity levels and rate of genetic drift) of the next generation (Waples 2022, 2024). N_e is an area with rapid developments ongoing in terms of theory, simulation and collection of field data, and there is an ongoing flux in the understanding of the set of inter-related topics that fall under its umbrella (e.g. Allendorf et al. 2024, Kardos & Waples 2024, Laikre et al. 2016, Ryman et al. 2023). For example, there are slightly different forms of effective population size, referred to as inbreeding, variance, additive genetic variance, linkage disequilibrium, eigenvalue, coalescent, local, global, and metapopulation N_e (Ryman et al. 2019). Each represents a different, but related, concept, and each can be calculated in different ways from different data such that caution is needed when extracting operational values from the literature.

The most important point is that, as N_e increases, the first benefit is a reduction in the probability of short-term **inbreeding** which is critical to maintain demographic viability (because inbreeding is often associated with reduced fitness, Liberg et al. 2005). However, conserving the full genetic variation within the population and allowing space for new variation to arise is essential to maintain the adaptive and **evolutionary capacity** of the population over longer time scales. This typically requires much larger population sizes. For several decades a rule-of-thumb has existed stating that an N_e of 50 is necessary to avoid inbreeding and an N_e of 500 is necessary to maintain the evolutionary potential of the population. Although this **50:500 rule** was developed in the 1980's from a combination of domestic animals data, lab animal data and theory it has remained widely used in the absence of a better rule of thumb. While noting that there have been calls to upgrade it to a 100:1000 (Frankham et al. 2014, Rosenfeld 2014, Traill et al. 2010), it is the 50:500 rule that has been recently accepted as an indicator for monitoring the genetic health of populations as part of the Global Biodiversity Framework (GBF) (Hoban et al. 2020, Mastretta-Yanes et al. 2024). Other versions include the potential use of a 100:500 rule to exercise precaution on the timescales that influence current policy cycles.

Any such rule-of-thumb obviously glosses over myriad details caused by differences in species' life-history, ecology and management, and as such should only be considered a rough guide (Hoban et al. 2024). This discussion over the values (50:500 vs 100:1000) merely reflects the fact that maintaining long-term evolutionary potential requires very large populations and that populations **cannot be too large, or too connected**! It should be noted that increased viability is enhanced by the degree of connection between populations as well as the size of a population.

Calculating the N_e of a population is not a trivial task as it is not simply the number of (potentially) reproducing or adult individuals. It is possible to **calculate it directly** using a variety of genetical methods (Sindicic et al. 2013, Skrbinsek et al. 2012, Snjegota et al. 2021), although there are many potential challenges and pitfalls that need to be carefully avoided (Kardos & Waples 2024, Ryman et al. 2023). Interestingly, such methods also work on historical, or even zooarchaeological, material permitting the reconstruction of long-term changes in N_e over time (e.g. Rodriquez et al. 2011).

In most cases N_e needs to be **estimated indirectly**. The total size of the population is normally proportional to the effective population size. One important consideration is that formally speaking in calculations of N_e it is normal to only consider the number of mature or potentially breeding adults (of both sexes), a parameter that is known as **census population size**, or N_c. Because most field monitoring methods quantify other parameters (see Box 2), it is often necessary to use **conversion factors** to calculate N_c from the metrics obtained in practice (Mergeay et al. 2024).

Depending on species life histories the ratio between census population size and effective population size (known as the **N_e/N_c ratio**) varies dramatically. For example, many species of marine fish that produce eggs in massive numbers will typically have very low, and often highly variable, ratios. Species such as large carnivores tend to have less variable and larger ratios, typically in the range from 0.1 to 0.4 (Clarke et al. 2024, Harris & Allendorf 1989, Hoban et al. 2020, Mergeay et al. 2024). For example, in cases where the N_e/N_c ratio is 0.1, it would mean that the effective population size is 10% of the census population size (mature individuals). Many large mammal models use an assumed default value of 0.2 or 0.25 (e.g. Dussex 2024, Waples 2022, 2024).

Effective population size can also be influenced by the way a species is managed. For example, different harvest / culling / control strategies or mortality patterns may influence the parameter depending on how they influence the variation between individual reproductive success or select for specific properties.

There is one important additional consideration about genetics. Effective population size only reflects the degree of heterozygosity in a population and does not measure the **allelic diversity**. Allelic diversity is the real foundation for long term evolutionary adaptation, and is unfortunately far more sensitive to bottlenecks. It is therefore important to be aware that N_e does not tell the whole story of the genetical health of a population (Allendorf et al. 2024). Conserving allelic diversity is best done by conserving the widest range of surviving populations and sub-populations, especially those for example that result from different subspecies, or come from different colonisation routes, or survived in different glacial refuges, or occupy different ecosystems (Carroll et al. 2020, Swenson et al. 2011).

With respect to setting goals for population recovery the key point is that in the short term populations have **to urgently reach an N_e of 50 and that long-term conservation requires an N_e of at least 500**. Assuming an N_e/N_c ratio of 0.2, for example, this would translate into census population sizes (mature individuals) of 250 and 2500, respectively. It is also urgent that all surviving source / **relict populations** are conserved so that their genetic diversity can be included into the pool from which future populations can draw from as they adapt to the increasing rapid rates of environmental change.

In practice reaching these goals is going to require the contribution of populations that stretch across many borders, including international borders, such that **connectivity within and between populations** is the essential goal. Connectivity is an important issue when considering the fragmented nature of the European landscape and much knowledge is available with respect to barrier effects, and mitigation measures, for the effects of roads, railroads etc. However, a new consideration concerns the unprecedented increase in border security **fencing** of the last decade triggered first by the migrant crisis and then by the development of war in Ukraine (Linnell et al. 2016). The current situation has effectively led to a near continuous border fence running along the eastern border of the continental EU with Belarus and Russia, in addition to an internal fence along the Hungarian-Serbian border, and external fences on the Turkish border. The fences on the Belarussian and Russian border will have dramatic effects on the overall level of connectivity between European carnivore populations and those in the larger populations to the east. The implication is that Europe can no longer count on geneflow from these populations and must therefore plan for viability in effective isolation. The planned border fences on the Finnish-Russian border are unlikely to have such large effects on Fennoscandian geneflow because they are only planned to cover shorter sections of the border. Veterinary cordon fences designed to impede the spread of African Swine Fever in wild boar are an additional obstacle, with thousands of kilometres of fencing appearing in some European countries with little environmental impact assessment.

Box 2 Linking population monitoring data obtained in the field with key assessment concepts.

Approximate estimates of effective population size (Ne) are typically calculated based on the number of mature individuals / adults / potential breeders in the population. This is known, somewhat confusingly, as the census population size (Nc) by geneticists. However, very few, if any of the field census methods in use for large carnivores actually directly measure this parameter. Different large carnivore species are typically monitored in different ways in different areas depending on their behaviour and ecology, local climate conditions, available resources, and the extent to which different stakeholders and institutions are involved.

The non-invasive DNA methods widely used for bears and wolverines (and sometimes wolves) typically estimate the total size of the population, i.e. animals of all age classes including young-of-the-year (N_{tot}).

Snow-tracking methods (wolves and lynx), natal den surveys (for wolverines), counts of female bears with cubs-of-the-year, and methods like howling surveys (wolves and jackals) typically record the number of reproductive events or reproductive groups like wolf packs or pairs (N_b). Such methods rarely produce statistical estimates of uncertainty.

Camera trapping can produce different values. For wolves, and lynx in many areas of northern Europe, camera trap data is mainly used to produce more observations of reproductive units / events (for lynx, wolves, bears) – and thus contributes to Nb. In central Europe, camera trapping of lynx is typically analysed with capture-recapture methods that can either produce statistical estimates of total population size (N_{tot}) or of the number of independent animals (Ni) if the visually recognisable dependent kittens (<1 year old) are excluded from the calculations.

Based on demographic data (birth and mortality rates for different age classes) it is possible to create conversion factors that allow a calculation between different values. There are examples of specific conversions, typically between N_b and N_{tot}, in regular use for wolves (Chapron et al. 2016), lynx (Andrén et al. 2002), and wolverines (Landa et al. 1998). It is important that these are locally adapted because different populations may have different demographic rates or different patterns of social structure. It is also critical to consider important practical details such as at what time of the year data is collected because mortality can be quite high among juvenile age classes. Hunting can also strongly influence numbers and age structure so it makes a difference in a census is performed before, or after, the hunting season.

Such calculations will require data on the tendency of individuals of different ages to reproduce. While this data is widely available for females of species like bear, lynx and wolverine it is much less available for females of golden jackal and wolves and for males of all species. There may be large differences between the number of males physiologically capable of reproducing and those that actually reproduce in polygynous species.

If effective population size is going to become a key benchmark for population assessment there will be a need to utilise the best available research and monitoring data to produce realistic, and comparable, conversion factors between what is actually censused in the field and the idealised value of N_c (mature individuals) that can be used to approximate Ne (effective population size).

An additional benefit of these harmonised conversion factors would be to make the presentation of population estimates more comparable between regions or countries (see Kaczensky et al. 2024 for a discussion of the problem).

5.3 Ecosystem Functionality & Representation

Ecological functionality is a key concept within the emerging recovery assessment frameworks. The Habitats Directive aim is to *" … contribute towards ensuring bio-diversity through the conservation of natural habitats and wild flora and fauna ….*" (Article 2(1)) and the definition of Favourable Conservation Status mentions that species should be " *…. A viable component of its natural habitats, …*" (Article 1(i)). Furthermore, the current definition of favourable reference range "*range within which **all significant ecological variations** of the species are included for a given biogeographical region*" further places a focus on species being concerned as interactive elements of their environment. The FRR definition from the Article 17 reporting guidelines is even more explicit than the Directive text as it focuses on all interactions in each biogeographical region. Bijlsma et al. (2019a) articulate this as involving the conservation of "*ecological / genetic variations within the (historical) range i.e. geographical, climatological, geological and altitudinal gradients as well as significant differences in historical landuse*".

Large carnivores are potentially strongly interactive species (sensu Soulé et al. 2005). These interactions can include;

- Behavioural, demographic and selective impacts on their prey populations.

- Numerical impacts on smaller carnivores (meso-predators).

- Interactions with each other.

- Providing carrion for scavengers.

- Seed dispersal and habitat modification (mainly bears).

However, the nature and strength of these interactions will vary dramatically between large carnivore species, with the structure of ecological community within which they live, with the degree of human impact on the landscape and with the overall environmental productivity of the ecosystem. Strong top-down effects on other trophic levels (also known as **trophic cascades)** may operate in some locations, but both theory and empirical data indicate that their occurrence and strength is likely to be highly **context dependent** (Hayward et al. 2019, Ray et al. 2005, Terborgh & Estes 2010). This context dependence makes it very hard to predict the strength of carnivore impacts, especially in human-modified landscapes where humans influence all trophic levels (landuse, hunting of shared prey, providing livestock and supplementary food mortality impact on the carnivores) (Kuijper et al. 2019, 2024). Although we have little data, it can also be expected that human induced mortality of carnivores may also influence their ecological functioning in addition to their demographics.

A large degree of focus is currently being spent on comparing ecosystem function to ancient, or so called "natural" ecosystem states with minimal human intervention. While this may be of scientific interest, it does not provide very useful guidance for future orientated recovery visions in continental scale landscapes that will always be heavily modified by human agency (Linnell et al. 2015) to the extent that they must be viewed as socio-ecological ecosystems (Levin et al. 2015), albeit with a wide variation of degrees of human impact.

It is also very hard to develop metrics to measure the degree of functionality. The most practical and basic measure of functionality is to document that the **structure of an ecosystem** has been restored (i.e. all of the strongly interacting and important species are at least present), even if the exact nature of the **dynamics** between them is not known. This would be to simply recognise the permanent **presence of reproductive populations of large carnivore**s in different areas as a metric for the potential for these functions to occur within that local area. The fact that the expression "*all significant ecological variations*" is defined within the context of range (FRR), rather than population (FRP) is a powerful argument that the intention is to view this as a qualitative goal rather than quantitative. The presence of large carnivores in different ecological settings will create the potential for the full diversity of ecological interactions to occur and also satisfy the need for representation of different settings. In a measurable sense this would involve ensuring the permanent presence and / or presence of reproductive units of large carnivores in;

- Parts of all of the **biogeographical regions** that can be considered natural range.

- All **Natura 2000 sites** designated for the species.

- Within all major **ecoregions** / broad **habitat** types / **topographic** formations.

- Within all of the different potential **prey communities** (or forage types for bears).

The extent to which the carnivores, their prey, and the prey habitat are directly influenced by humans should provide a proxy for the degree of ecological functionality based on the assumption that low human intervention increases the degree of large carnivore function. Clearly this is not a model that can be generally advocated for the whole landscape, but may be relevant for protected areas, including Natura 2000 sites, etc.

To ensure coherence with the ideas of redundancy, representation and resilience it would not be enough for one population of each species to be present in one biogeographic region somewhere in Europe. It would rather require that as many as possible regions are occupied, in other words, each member state should have an independent responsibility to contribute to this is the various ecological conditions present within their borders.

Our proposed focus on effective population size (see previous section) implies that there is a need for an even greater focus on ensuring that large carnivore populations achieve a high degree of connectivity on a continental scale. This focus, together with our proposal for incorporating a new focus on ecological functionality and representation (this section), will require the presence of large carnivores across very large parts of the European landscape. In these massive areas they will occur within a huge diversity of different settings with very different degrees of human landuse and activity, ranging from semi-natural protected areas to multi-use forests, agricultural and peri-urban areas. This has two implications. Firstly, is a need to develop realistic views of the extent to which carnivores can assert significant ecological impacts. In more human-dominated landscapes their impacts are likely to be masked by human effects on all trophic levels and will be far from "natural". In more natural areas the scope for larger ecological effects is greater. In other words, there is a need to create realistic expectations of very different ecological functions in different settings. However, because of the need to ensure connectivity, carnivore populations inhabiting areas where their ecological role may be diminished will still be essential for ensuring the much needed connectivity. Secondly, the need to allow large carnivores to occupy many areas that will have heavy human presence implies that their management in these areas must be conducted in a pragmatic manner that promotes tolerance and coexistence based on the insight that many of the "ecological functions" may also be a source of conflict for some stakeholders (see section 5.4).

5.4 Recognising the costs of living with success

In addition to an increase in the understanding of ecology and genetics of large carnivores, the last decades have seen a dramatic increase in the level of inter-disciplinary research and policy experience focusing on both the **impacts and conflicts** associated with successful large carnivore conservation. These are diverse and range from the **economic** (e.g. depredation on livestock, potential impact on harvestable game populations, attacks on companion animals) to the **social** (e.g. conflicts between different people over the appropriate way of managing large carnivores) (see Linnell 2013 and Redpath et al. 2013 for reviews). The last years have also seen the extent to which these conflicts have become **political** in nature in Europe (Niedzialkowski 2022, Zscheischler & Friedrich 2022, von Hohenberg & Hager 2022) and multiple member states have expressed a desire to change the annex designations of both wolves and bears on the directive. At the time of writing, the European Commission has taken active steps to downlist the wolf on the Bern Convention (December 2024) as a first step in a process to change its status on the Habitats Directive.

Such conflicts are not limited to large carnivores in Europe, and there is a strong global movement to both recognise them and adapt the ways of doing conservation to move to a more **socially just** form of conservation policy implementation (Levin et al. 2015, Milner-Gulland 2024, Redpath et al. 2017). The IUCN has consolidated these ideas in specific guidelines on Human-Wildlife Conflict and Coexistence (HWC)(Zimmermann et al. 2023), and the Global Biodiversity Framework now includes both a reference to HWC and an indicator to monitor actions dealing with them.

The implementation for European large carnivore conservation and the setting of FRVs must recognise that conflicts with large carnivore presence will be diverse and widespread, although highly variable in space and time, and the distribution of costs and benefits will be highly scale dependent (i.e. local costs and distant benefits). This means that from a conservation science best-practice perspective there will often be a need to consider socio-economic factors as well as ecological factors when planning for recovery, both in terms of range and densities. Article 2(3) of the directive requires the need to take into account "*economic, social and cultural requirements and regional and local characteristics*". Many national action plans for large carnivores are explicitly based on balancing conservation ambition with conflict (see Appendix 1 for carnivore examples and Van Eldik et al. 2024 for wider examples from other species). Many of these action plans have also been developed with extensive stakeholder involvement through elaborate participatory practices which are viewed as being best-practice in terms of ways to identify pathways to coexistence. However, this broad set of conditions has not always been reflected in guidelines on setting FRVs which have tended to focus on issues of technical feasibility without considering the socio-economic factors. The current state of research-based knowledge on this issue indicates that it is both practically and politically problematic to ignore socio-economic factors when considering FRVs for species groups like large carnivores that are associated with real economic costs, potential risks to human safety, and widespread social conflicts. This is a very special situation for large carnivores because of their very specific ecologies and complex relationships with people. Levin et al. (2015) express the need as "*It is a truism that if we do not know where we want to go, we will surely have a hard time getting there. Perhaps equally as axiomatic is the fact that if a broad constituency does not contribute to defining the destination, the road will be very bumpy*."

There are clearly a range of opinions concerning the manner in which these economic, social and cultural issues can and should be included in (1) setting FRVs, (2) setting targets that go beyond FRVs, and (3) impacting the way of achieving these objectives. It is also unclear where the border between technical aspects and economic, social and cultural aspects lies as well as the relevant importance of aspects in the directive text and various guidelines. Although the final CJEU judgement is not yet available, the opinion of the advocate general in case C-629/23 would seem to indicate their admissibility as long as the criteria of FCS are achieved. Finally, Article 191(3) of the TFEU would also seem to provide some openings for "*the economic and social development of the Union as a whole and the balanced development of its regions*" being taken into account.

There is also a need to make a distinction between the extent to which these factors can influence setting FRVs at different levels. These guidelines propose FRVs at the population level based on biological criteria (effective population size and connectivity) that cannot be compromised without endangering the objectives of the Habitats Directive. There is also a need to ensure that the contributions of all member states sharing a population add up to a population level FRV to reach this biological threshold. However, there is much more scope at the sub-national level to incorporate these social, economic and cultural factors to determine how much above these thresholds FRV values, or targets, are set.

Overall, there is clearly a need for clarity around these issues concerning both the interpretation of the Habitats Directive text and the relationship between different EU policy areas that may be impacted by large carnivore conservation.

5.5 Recognising the diversity of European countries capabilities

In the last 16 years since the 2008 guidelines were developed there have been considerable **positive developments** in virtually all European large carnivore populations. Most notable has been the expansion of the wolf population in Central Europe. From 2008 to 2022 the number of wolves in Germany increased from

5 packs to 185 packs and the first wolves arrived in Denmark, the Netherlands and Belgium in 2012, 2015 and 2018, and then bred in 2018, 2019 and 2019, respectively. Non-resident wolves have even been recorded in Luxembourg since 2017. Reconnection has been established between many previously isolated populations such as the Alpine, Dinaric-Balkan and Carpathian populations (Boitani et al. 2022). Bear populations have also increased, for example in the Cantabrian mountains of Spain, the Pyrenees, and the Italian Alps (Kaczensky et al. 2024), and multiple lynx reintroduction / reinforcement projects have been conducted, or are underway, in continental Europe.

These positive developments, especially the return of wolves to some of the smallest and most heavily human developed countries (Box 3), require for the first time a consideration of what concepts like FCS and the associated FRVs mean for **very small countries**, or very densely populated countries, which may not have the best preconditions for large carnivore conservation.

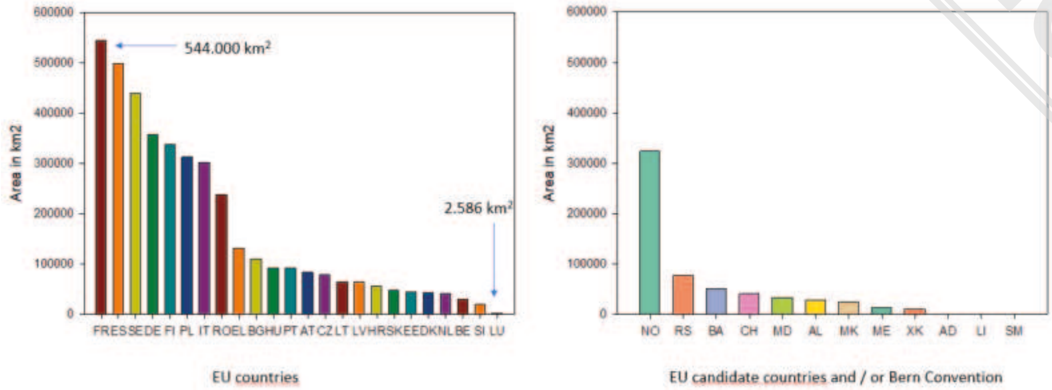
Size *per se* has not been an obstacle to making large contributions to large carnivore conservation as small countries like Slovenia and Estonia have long hosted substantial populations of bears, wolves and lynx. However, these countries are heavily forested and have low human densities (and bears are provided with supplementary feeding in many cases). The situation for the Benelux countries, for example, is very different due to the combination of small size, low area of forest cover, and high human densities. There is no legal doubt (especially after CJEU ruling on Case C-601/22 in 2024 concerning Austria) that **all countries have an obligation** to contribute to the objectives of the Habitats Directive. But the question remains if it is reasonable (proportional) for such small countries to have the same absolute requirements for making a contribution to EU objectives as large countries. In other words, would Luxembourg have the same absolute requirement for FRVs as France which is 220 times larger? If so, then many (most) countries would never be able to achieve FCS for large carnivores using any reasonable definition of the concept. An alternative interpretation would consider that a countries expected contribution would be scaled to their size or environmental preconditions, and that their assessment of FCS would then be relative to what they could maximally contribute. This concept was originally proposed by Epstein et al. (2016), but has not been explicitly developed or addressed since. However, the fact that potential habitat / distribution maps are suggested as a possible means of setting FRVs would indicate an implicit understanding of this interpretation because such maps are by definition adjusting to the local realities within member states.

A still contested aspect concerns the extent to which social, cultural and economic aspects should be given weight in this evaluation of a countries potential contribution. One example concerns Sami reindeer herding in northern Fennoscandia where the difficulties of preventing large carnivore depredation, and the paucity of wild alternative prey, have led management authorities to adopt relatively low levels of recovery ambition for large carnivores, especially wolves, in an area that constitutes almost 40% of the countries' area (Rasmus et al. 2020). Similar claims have been raised by European sheep farmers in some areas (see CJEU Case C-601/22). The ruling in the latter case implies that member states have some discretion in evaluating this issue, but the extent of this discretion is likely to be contested. The previous section (5.4) raises the need for legal clarity around this matter.

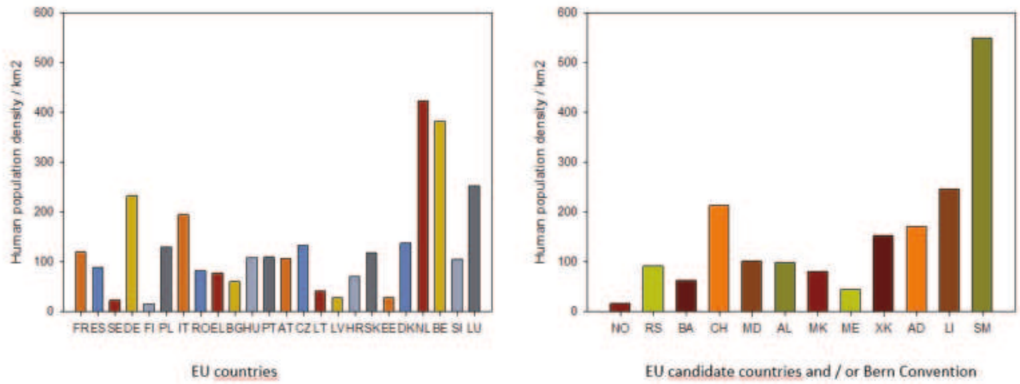
Box 3 Diversity of European countries preconditions for large carnivores

In a European context large carnivores are present, or potentially present, in 24 EU countries, in addition to 12 other countries that are either EU-candidates, potential candidates, or associated countries (European Economic Area) and bound by the Bern Convention. As such, all are bound by similar pan-European legislation, but have very different preconditions to contribute to these common obligations.

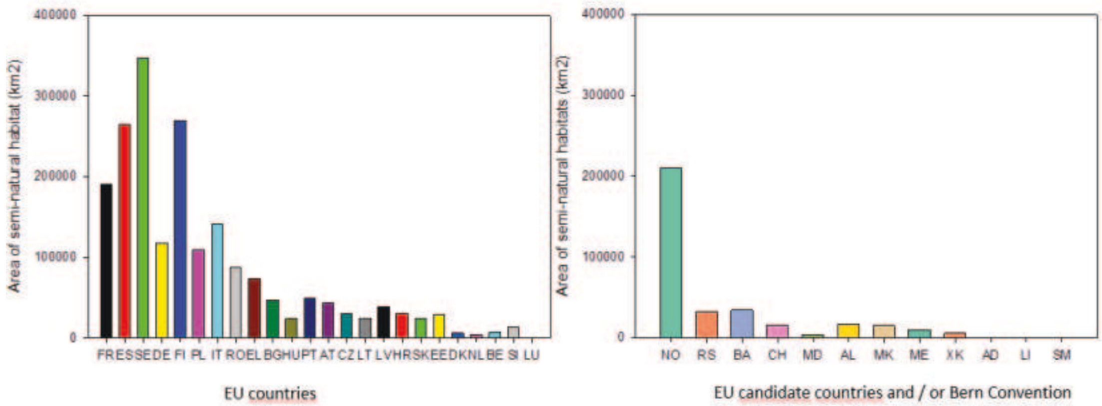
Size of the countries – varies by factor of 220.



Human population density – varies by factor of 23.



Semi-natural habitats – varies by factor of 352 in EU.



Estimating the potential habitat for large carnivores is a complex task as it depends on many factors such as human density, amount and configuration of habitat, topography, amount of linear infrastructure, many details of human landuse etc (e.g. Cimatti et al. 2021, Cretois et al. 2021, Cristescu et al. 2019, Magg et al. 2016, Oeser et al. 2023, Scharf & Fernandez 2018). However, for the sake of a quick illustrative comparison we have presented graphs with the area of semi-natural habitats – extracted from the EU's EUNIS classes Level 1 data (Weiss & Banko 2018) – using classes D (mires, bogs and fens), F (heathlands, scrub and tundra) and G (woodland, forest and other wooded land). Grasslands were excluded because they only constituted a minor area in this dataset. The figure shows that the absolute amount of suitable habitat varies by a factor of 352. Based on the figure we can recognise at least three classes of country. The very small with less than 10.000 km^2 of semi-natural habitats (Luxembourg, Netherlands, Denmark, Belgium), medium sized with from 10.000 km^2 to 50.000 km^2 (Slovenia, Estonia, Latvia, Lithuania, Slovakia, Croatia, Czechia, Austria, Portugal, Hungary and Bulgaria), and the large with everything from 50.000 km^2 to 346.000 km^2 (Greece, Romania, Italy, Poland, Finland, Germany, Sweden, Spain, France). Again, it should be underlined that these figures are intended to be illustrative, and any application of our guidelines would require much more robust habitat suitability models and a more objective evaluation of cut-offs between size classes of country. The main take-home message is simply that different countries have radically different preconditions for conservation which requires scaling ambition to these preconditions.

6 FRVs vs targets

There is no dispute that the objective of the Habitats Directive and other nature conservation legislation is to both prevent extinctions and to promote recovery **beyond minimal levels** (Mehtala & Vuorisalo 2007). Extinction is a clearly defined state in ecological terms and there are clear scientific frameworks to set thresholds for avoiding short-term extinction (e.g. PVA approaches). In contrast, **recovery is not a clearly defined state** in ecological terms and there are no undisputed scientific tools to determine thresholds. As a result it is clear what we are trying to avoid, but less clear as to what we are trying to collectively achieve. The recovery of large carnivores will inevitably be associated with significant conflicts with diverse stakeholders and economic interests that will vary between countries and regions depending on environmental, economic, social, cultural and political factors. It is therefore likely that different jurisdictions will have different motivations and different capabilities on how far they wish to pursue the process of recovery.

As a result there may be a point at which member states (or sub-national units with delegated authority) will switch the question of recovery ambition from being one of "what level of recovery **do we need** to satisfy biological needs and legal obligations to the EU" to "what further level of recovery does our society **wish to live with**"? Along this gradient of recovery there will also be a switch in emphasis as different issues, processes and components of recovery get greater or lesser emphasis (Figure 1). This issue has been previously discussed, albeit indirectly, in numerous guidelines where it has been stated that FRVs are not the same thing as targets. However, for the purposes of going further we think **it would be helpful to view FRVsas as the point at which obligations to the EU's collective effort to cooperate on conservation end (i.e. achieving and maintaining FCS), and where there is a greater opening for national, or sub-national, democratic processes to decide on higher levels of ambition.** This implies that for the purposes of these guidelines we recommend that FRVs should be viewed as achievable targets that member states can realistically reach. Member states will of course have the freedom to set even more ambitious goals for themselves.

There has been much discussion as to whether FRVs are meant to be the same thing as targets. In some interpretations FRVs have been described as long-term, ambitious "stretch goals" that may be hard to achieve in practice, but which provide a long term aspiration that members can strive towards. This interpretation is, however, rather complicated as multiple legal rulings have indicated that a member state's management options (under both Annex IV and Annex V designations) are constrained until FCS (which requires achieving FRVs) is reached. Therefore it is practically significant to have FRV values that can be reached and maintained, otherwise management options will be permanently constrained.

For species on Annex IV this difference may t be less crucial because the limitations on killing resulting from Strict Protection will likely set limits on how much constraint on further population growth can be imposed. For species on Annex V it will be more crucial as reaching and maintaining FCS is the only constraint imposed on member states' freedom of management and is an essential yardstick against which adaptive management can be measured.

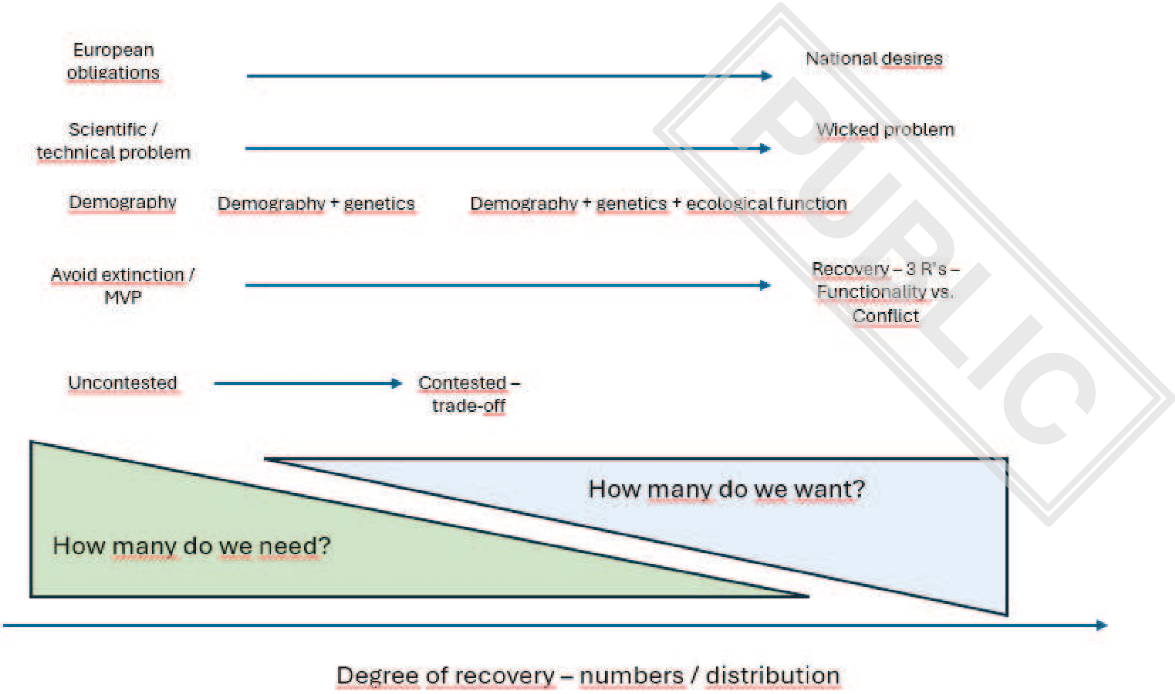


Figure 1. illustrates the way that questions and concepts vary along the recovery gradient. As recovery proceeds the questions will change from "how much recovery do we need?" to "how much recovery is wanted", the role of EU obligation will fade in favour of the role of national desires, and different concepts will gain in importance. Favourable Reference Values represent the point at which obligations to the EU switch to being national desires.

B: Practical approaches for setting Favourable Refence Values

7 Linking biological concepts and the terminology of the directive

In this section we link the legal / administrative terms understanding of FRVs with the scientific concepts that have been discussed in the previous sections and propose practical approaches to make these connections.

7.1 Scales of assessment

A central question concerning reporting of FCS and setting of FRVs has concerned choosing the appropriate scales of assessment. The default level within EC documents has been at the level of each **biogeographic region** within each member state. During the last decades there has been a growing realisation that different species have such different ecologies that they need different scales of assessment. For example, this concerns migratory species as well as species that make very large movements or occur at low densities such that international (transboundary) coordination is required to assess and conserve their populations. Bijlsma et al. (2019a) cover the issue of scale at great length, presenting a detailed breakdown of different spatial structures and movement patterns for species and aligning this with the most appropriate scale of assessment. Large carnivores clearly fall into the category of species that generally **require a supranational scale of assessment**. The Article 17 reporting guidelines currently open for joint assessment of shared populations, although there is still a need to disaggregate the information down to the biogeographic regions and member state level. These developments represent a realisation of the importance of considering species ecology and life-history, although there is still some way to go to fully operationalise this within reporting structures (i.e. this is currently included under “additional information”).

The 2008 guidelines (Linnell et al. 2008) strongly encouraged the adoption of a **population approach** for large carnivores, and this has been followed up in expert led reporting processes and evaluations ever since (see Boitani et al. 2015, 2022, Kaczensky et al. 2013, 2021, 2024). Many member states have embraced the idea in principle and there has been an increased degree of technical cooperation in research and monitoring. Regional conventions (Alpine Convention and Carpathian Convention) and initiatives (Dinaric-Balkan-Pindos Large Carnivore Initiative, Central European) have been promoting transboundary coordination and made many gains in terms of information exchange, dialogue and practical coordination. A working group within the Alpine Convention came as far as proposing a model for commitment between parties (Schnidrig et al. 2016). Unfortunately there has of yet been no signing of politically binding population level management plans being developed (see also section 4).

Based on a combination of actual distributions, genetics, history, topography and administrative / political borders the LCIE has proposed an operational set of populations for each of the large carnivore species, for a total of **35 populations**, plus a number of recently established “occurrences” (Table 2). The best available knowledge currently recognises 10 bear populations, 11 lynx populations (plus 2 small occurrences that have not yet stabilised enough to be called populations), 9 wolf populations, 2 wolverine populations and 3 golden jackal populations (with many small additional and recently established occurrences scattered across northern Europe). 29 of these 36 populations are transboundary (Kaczensky et al. 2024) involving from 2 to over 10 countries. Some countries find themselves at the interface between multiple populations. For example Poland contains portions of 3 wolf populations (Baltic, Carpathian, Central European) and 3 lynx populations (Baltic, Carpathian), and Austria has documented the presence of individual wolves from 4 different populations (Alpine, Dinaric-Balkan, Central European and the Carpathians).

A high degree of continuity in distribution and / or sufficient migration to have a demographic effect was a prerequisite for aggregation into a population, although some more or less continuous distributions were split to create more homogenous operational units, or management units. It should be noted though that the situation on the ground is dynamic (and some borders are contested, e.g. Gula et al. 2020, Szewczyk et al. 2019). As populations expand and reconnect there may be advantages of merging some of them into larger units if this facilitates coordination. However, it is also possible that future research will reveal a greater than anticipated degree of sub-structuring or unrecognised barriers such that what appears to be single populations today may actually need to be split. In practice there will be an optimal size for units, big enough to be biologically meaningful, yet small enough to be manageable. There are also limits to the practical size of management units. At too large scales there are simply too many actors, stakeholders and authorities involved to make management possible. Furthermore, even for wide ranging species there are limits to the size of a distribution that can be considered as a functional unit.

One unresolved issue concerns the **minimum threshold of movement** between areas that is necessary to recognise it as a functional genetic population. Based on our conceptual definition of a population there are both demographic and genetic aspects at play. For genetics it is usually recognised as being sufficient with 1-5 effective migrants per generation (i.e. animals that both move and reproduce), which in practice means a substantially greater number of animals making the move. Technically, the ratio of effective migrants to total number of migrants follows the N_e/N_{tot} ratio, so if this is 0.1 for example (Mergeay et al. 2024) then it would need 10 migrants to produce one effective migrant. Certainly anything less will be too little to justify aggregation, and the precautionary principle would indicate a need consider a higher rate of migration for this key parameter because of the massive impact of the fate of individual wolves on the outcome. A multi-year average will also be more robust than a single year's data.

A result of these considerations is that there are some cases where it might be necessary or justified to consider different units for assessment of effective population size than the populations that have been used since 2008. This is especially true for wolves, where Mergeay et al. 2024 have argued for example that the Italian and Alpine wolf populations should be considered as a single genetical unit, as could the Dinaric-Balkan and Carpathian wolves. Similar arguments have also been proposed for the Scandinavian and Karelian wolf and wolverine populations, the Alpine and Dinaric-Pindos bear populations and some of the central European lynx populations. This different scale of assessment for effective population size assessment vs the previous populations / management units from 2008 makes sense because of the different focus on genetics vs demographics, and was anticipated in the 2008 report. However, any such merging requires a serious documentation of actual exchange rates of animals or of real continuity of distribution.

Table 2. Currently recognised “populations” of large carnivores in Europe (Kaczensky et al. 2024). The two marked with an asterix are designated as “occurrences” implying that it is too early in their establishment to determine if they have become self-sustaining populations yet.

Population	Countries	Population	Countries
Brown bear		**Eurasian lynx**	
Alpine	IT, CH, AT	Alpine	IT, FR, CH, AT, SI
Baltic	EE, LV	Balkan	AL, MK, XT
Cantabrian	ES	Baltic	EE, LV, LT, PL
Carpathian	PL, CZ, SK, RO, RS, HU, UA	Bohemian-Bavarian-Austrian	DE, CZ, AT
Central Apennine	IT	Carpathian	PL, CZ, SK, RO, RS, BG, UA
Dinaric-Pindos	SI, HR, BA, RS, ME, AL, MK, XT, EL	Dinaric	SI, HR, BA
East Balkan	BG, EL	Harz	DE
Karelian	FIN, NO	Jura	FR, CH
Pyrenean	FR, ES	Karelian	FI
Scandinavian	SE, NO	Scandinavian	SE, NO
		Vosges-Palatinian	DE, FR
		Black Forest – Swabian Jura*	DE
		Pomeranian*	PL, DE
Wolf		**Wolverine**	
Alpine	IT, FR, CH, DE, AT	Scandinavian	SE, FI, NO
Baltic	EE, LV, LT, PL	Karelian	FI
Carpathian	PL, SK, RO, RS, CZ, HU, UA		
Dinaric-Balkan	SI, HR, RS, BA, ME, Al, MK, XK; EL, BG	**Golden jackal**	
Central European	PL, DE, CZ, DK, NL, BE, LU, AT	Continental	EL, MK, XK, AL, RS, BA, MD, ME, PL, DE, IT, RO, BG, HU, AT, CZ, LT, LV, HR, SK, EE, SI, UA (DK, NL, NO, FI, CH, ES, FR, DE)
Iberian	ES, PT	Samos	EL
Italian peninsula	IT	Peloponnese	EL
Karelian	FI		
Scandinavian	SE, NO		

7.2 Favourable Reference Population

Because of the scales at which large carnivores occupy space (large individual home ranges, low population densities) it is only on the scale of the **population** that any reasonable **long-term viability** can be achieved – and even then really long-term genetic viability will depend on interconnections at the metapopulation, or continental wide, level. As a consequence, the only biologically meaningful scale of assessment is at the population level. However, it is at the **political level** of the member state that legal obligations and management structures lie.

This implies that we need to view all assessment criteria (FRVs) on **two scales**, following Bijlsma et al. (2019a) suggestion of adopting "partial FRVs". On one level FRVs are properties of the biological populations, while on another they are properties of the administrative / political scales of the member states. We propose to use the subscripts "POP" for the population level and "MS" for the member state level – such that FRP at the population level becomes FRP_{POP} and at the member state level it would be FRP_{MS}. Similarly, we would have FRR_{POP} and FRR_{MS}. A natural consequence of this is that the same concepts, FRP and FRR, would take on different meanings and have different values at different scales.

At the **population level** we propose that these concepts need to be **anchored on absolute values** because they must ensure the long-term viability of the species concerned, and species life-histories are not scaled or adjustable to political realities. Based on the ideas initiated in the 2008 guidelines, but further refined above, we recommend that the ideal default benchmark for FRP at the population scale be **an effective population size of at least 500** (i.e. $FRP_{POP} = N_e > 500$). If we use the N_e/N_{tot} ratio proposed by Mergeay et al. (2024) then this would correspond to 5000 wolves or 500 packs (other ratios from the literature would of course give other conversions, for example 0.2 would result in half the number). In practice, this benchmark is broadly similar to that of IUCN criteria D which was one of the options included in the 2008 guidelines, although it represents an increase on the IUCN criteria E option. The change is justified on multiple arguments, including;

- Clarification on the meaning of "long-term" in the Directive text to embrace long-term genetical aspects of viability (i.e. closer to evolutionary time than decades).

- We now focus on a single metric (rather than Criteria D or E that may gave different answers) that can be directly calibrated to species ecology.

- Developments in the policy related, conceptual, and scientific understandings of recovery (i.e. spoltlight on Habitat Directive goals being more than just avoiding extinction).

- Developments in the understanding of the importance of genetic viability for long term survival of species.

- New empirical studies on the impacts of inbreeding and the observed effective population size of various European populations.

- The recent disconnect between continental European and Russian / Belarussian populations due to border fences.

- The benefits of harmonising concepts and reporting to the Directive with that of the Global Biodiversity Framework.

Because of the importance of **non-EU countries** to many of the populations (especially in the Alps, Scandinavia, Dinaric-Balkans and Carpathians) we recommend that population level assessments should be able to include the individuals in these non-EU countries because they are bound by the Bern Convention (and Bonn Convention) that harmonises with the broad objectives of the Directive (Eriksen et al. 2020) whenever possible. This follows on existing recommendations with the Article 17 reporting guidelines. The CJEU has clearly stated that the Fennoscandian countries (and presumably the Baltic countries by extension) cannot count on their connectivity with Russian populations in an *a priori* manner, because Russia is not bound by the same legislation. However, if research and monitoring can document that there is an actual and ongoing geneflow across the border it would seem disingenuous to deny the reality of the situation (i.e. if the parts of the population that extend into Finland, Sweden and Norway or the Baltic States are really extensions of a continuous population with large effective population size). We would therefore suggest that this connectivity be counted as long as it can be continuously documented.

By the same logic, it should be possible to at least partially **consider inter-population connectivity** (i.e. metapopulation level effective population size) within the area covered by the Habitats Directive and Bern Convention if it can be documented that it is of significant magnitude when assessing population level FRP. The implication is that we may need to consider some of the larger meta-population structures as units rather than the current populations where there is a high degree of connectivity. However, it would be completely against the forward looking spirit of the directive if the reestablishment of connectivity between populations was subsequently used to downsize the level of conservation ambition / obligation from that which was present before reconnection.

On the **member state level** there needs to be a certain degree of **scaling** of concepts to recognise the different sizes, suitable habitats, and other realities of the diversity of countries that share populations. This means that the concepts become more relative than absolute as long as they sum up to a value that brings the shared population to a level consistent with its FRP_{POP}. Ideally, member states would scale their contributions in a manner which is proportional to their size and presence of habitat (while considering social, cultural and economic aspects to the extent which is legally possible). Where member states host more than one population the cumulative impact of contributing to all of them should be considered so it is their **total contribution** to large carnivore conservation that counts.

Member states have **diverse situations** (Box 3). It is therefore natural that they should have a certain degree of **flexibility** when it comes to setting their national contributions (FRP_{MS}) to the overall population level viability (FRP_{POP}). There are multiple approaches currently in use (see Bijlsma et al. 2019a, and Appendix 1 in this report) that could be considered.

- Filling up all, or a substantial part of the **potential available habitat** would be a very ambitious objective, although the amount that can be filled will differ in many contexts because of conflicts with landuse and rural communities. Defining potential habitat a priori for golden jackals and wolves can be a challenge because of their high degree of ecological flexibility and tolerance for anthropogenic landuse (compared to lynx and bears for example), but for all species it is possible to identify gradients of preferred habitats or use semi-natural vegetation types (e.g. forests / heaths) as a very broad proxy. Carrying capacity is also hard to assess for large carnivores as it depends to a high degree on the way people manage the wild and domestic prey base (and forage) on which large carnivores feed. There is also a subjective element involved when opting for a “substantial part”. Epstein et al. (2016) suggested **50% as a minimum**, as does the “Nature-needs-half” movement (Müller et al. 2000). The 50% value is intuitive as it means that the status is then closer to full recovery than to extinction, even if it lacks an objective basis. Recent global initiatives to protect 30% of the planet's land area offer another potential benchmark. Either way, the choice will always be subjective (Svancara et al. 2005). Overall such approaches are probably most suited to the smallest countries with the least amount of potential habitat because of the challenge of realistically benchmarking other more absolute ecological goals to their small size.

- For medium sized countries a range of options for setting numerical values of FRP_{MS} may be appropriate related to **MVP concepts** (extinction risks less than 5 or 10% over 100 years) or preferably effective population sizes that at least **avoid inbreeding** on shorter time scales (i.e. N_e above 50, or 100 if applying precaution, but less than 500). Benchmarking to such concepts implies that member states are making real contributions to the overall population level viability and that even in isolation their portion of the overall population has a certain degree of viability on short to medium time scales.

- For larger countries it would be in keeping with a spirit of fairness and proportionality to **go beyond MVP or N_e >>>50** if their contributions are going to be proportional and if the overall population is going to reach an absolute benchmark linked to a FRP_{POP} with an $N_e > 500$. It is likely to be the need to reach this overall goal for FRP, as well as FRR (see below) that set the values for these countries. Dependent on habitat availability it would seem logical that many large countries should aspire to values closer to 500.

- There is a clear need for a degree of proportionality and **fairness** between countries such that they make contributions broadly in line with their potentials of size and habitat.

A special case concerns the **very small populations** of bears (Cantabria, Pyrenees, Apennines) and lynx (all populations apart from the Scandinavian, Karelian, Carpathian and Baltic). In many cases entire populations are present within a single country (such that $FRP_{POP} = FRP_{MS}$) or only shared by two. Most of these are very far from being able to reach a FRP_{POP} >500. Due to ongoing reintroduction projects many of the lynx populations have the potential to reconnect into large clusters which collectively may be able to reach this goal at a metapopulation level. Unfortunately these 3 bear populations are too geographically isolated to expect reconnection, and population growth occurs too slowly to reach such ambitious goals, on any time scales measured in decades. However, the importance of the Cantabrian and Apennine bears (and the Balkan lynx which is currently outside the EU) far exceeds their numbers because they represent unique genetic lineages / relics that are potentially of great importance to giving future bear populations the greatest possible genetic platform for adaptation. As a result it is important that conservation efforts are prioritised in these populations even if the current situation is far below any FRVs.

- For some of these cases it may be appropriate to **set FRP_{POP} with an operator** (>>CV) rather than a number. This would require members states to continue monitoring (both population size and the genetics) and achieve population growth. It may be realistic to set shorter term targets that aim to reach an N_e greater than 50 as soon as possible as a "**stage goal**" to mark progress on the long-term route towards more substantial recovery. Translocation of animals from other populations (assisted connectivity) could also quickly allow an increase in effective population size.

7.3 Favourable Reference Range

In the 2008 guidelines, Favourable Reference Range was somewhat superficially treated as being large enough to embrace the Favourable Reference Population and attempting to ensure connectivity. Many of the arguments from the previous section on FRP indicate a need to adopt a more rigorous and specific definition of the FRR. As for FRP, we consider that FRR needs to be considered at multiple scales, mainly that of the population and that of the member state, designated with the FRR_{POP} and FRR_{MR} subscripts, although conceptually the two levels are much more aligned conceptually. In addition, it may be necessary to consider FRVs at sub-national levels within federal states.

Firstly, the recommendation to benchmark population level Favourable Reference Population on effective population size (and ideally greater than 500, $FRP_{POP} = N_e>500$) implies that rather large numbers of animals are required because an $N_e>500$ implies that there will several thousand animals in the total population depending on the conversion factor chosen. This will require a large amount of space in all member states to give enough place for the **total population**.

Secondly, when taking long-term genetic aspects seriously it becomes apparent that maintaining **connectivity** is essential. This connectivity refers to all scales, both within a member state and between member states, to ensure internal connectivity within populations, and where possible between populations to create a functional metapopulation. There may be some acceptable discontinuities in distribution within these ranges as long as they are within the species' dispersal capabilities and not associated with unmitigated or impermeable barriers (highways, fenced railways, veterinary fences, border fences etc). As outlined in the 2008 guidelines, broad distributions with high connectivity are far more important than localised high densities in building long-term viability and resilience.

Thirdly, the FRR should embrace all of the **remnant distribution areas** of large carnivores such that the maximum possible range of genetic variation lies at the foundation of further recovery.

Fourthly, ensuring a wide range is the primary way to ensure that the necessary aspects **of ecological functionality** ("all significant ecological variations") are achieved. This implies that to ensure functionality, in addition to both representativeness and redundancy, FRR must include a permanent presence (equivalent to "Present regularly" (PRE) in Article 17 reporting terminology) in;

- The **Natura 2000 sites** designated for the species.

- At least part of all the **biogeographic regions** within the country.

- In all suitable **major ecosystems** and **prey communities** within the country.

The result of this need to take range seriously is that large carnivores are going to need to occupy very large amounts of space, with at least some species occurring in a high proportion of the European landscape. Their successful conservation will in effect mean that they will become "normal" (widespread) parts of the countryside. However, none of these objectives is strictly numerical leaving member states with certain **discretion** concerning how much of each biogeographic region needs to be occupied for example.

7.3.1 Natura 2000 in context

Until recently there has been relatively little continental scale focus on leveraging the full value of Natura 2000 sites for large carnivore conservation. Overall concerns have been based on the mismatch between their size, and the spatial needs of large carnivores (Boitani & Linnell 2015, Santini et al. 2016). Recent research has shown that some protected areas (which are included in the Natura 2000 network) actually have major importance for the persistence of some populations, for example lynx in the Bavarian-Bohemian forest system (Magg et al. 2016, Müller et al. 2014) and small bear populations in Cantabria, the Apennines and the Alps. Other studies have also identified that the Natura 2000 network can protect some key habitats for large carnivores (Diserens et al. 2017, Marucco & Avanzielli 2022, Santini et al. 2016, Votsi et al. 2016), although it is important to differentiate between those sites that are only Natura 2000 sites and those that are an overlay between Natura 2000 and other, more strictly protected, area types (Cristescu et al. 2019). Natura 2000 designation holds the potential for strong habitat protection that can benefit large carnivores (see CJEU court ruling C-404/09 with respect to Cantabrian bears) although the extent to which member states follow their responsibilities is variable (Sazatornil et al. 2019).

It is, however, important to realise that no protected area network will ever be enough to conserve large carnivore populations. Their long term persistence depends on their presence across very large areas of multi-use and human-dominated landscapes that will never be protected. Modern developments in conservation planning have shown the benefits of coordinating site-based (i.e. Natura 2000 and other protected areas) and whole-landscape based approaches to conservation to develop plans for connected landscapes (e.g. Hebblewhite et al. 2021 for a North American example). Leveraging the value of protected areas requires adopting a realistic understanding of their potential role. Due to the above mentioned scale mismatch there are few, if any, European protected area networks that will fully embrace substantial numbers of large carnivores solely within their boundaries. However, the extra protection that they afford individuals for all, or part, of their annual life cycle may be critical in some regions to provide a safe "core" from which expansion can develop (e.g. Müller et al. 2014). The rapid expansion of large carnivores has led to their colonisation of areas where no Natura 2000 sites exist for them because their future presence was not anticipated.

On the one hand the large carnivores may make contributions to the ecology of the protected areas through their ecological role as top-predators, although as highlighted above this may be highly contextual. Protected areas, including Natura 2000 sites, may represent areas where large carnivores can be allowed to display a greater degree of ecological functionality than in unprotected sites (Kuijper et al. 2019, Ordiz et al. 2013). On the other hand it is important to still bear in mind the nature of European protected areas. While some are managed as minimal intervention sites, the vast majority represent mosaics of natural and anthropogenic habitats (e.g. heath lands, meadows), where many of the anthropogenic habitats are themselves subject to protection under the Directive. Most protected areas contain human settlements and active forms of landuse such as forestry, livestock production and hunting (Tsiafouli et al. 2013, van Beeck Calkoen et al. 2020). Protected areas can also be associated with high rates of recreational or touristic visitation, which may be a driver of conflict in some cases (Penteriani et al. 2017). It is therefore important to adopt a realistic level of expectation into what the Natura 2000 network means for large carnivore conservation and what they mean for the Natura 2000 sites. Maximising the benefits and minimising potential conflicts requires careful management planning and the coordination of different policy instruments.

7.3.2 Biogeographic regions in context

Although the biogeographic regions are not mentioned in the directive's text, following current guidelines member states are assessed on species conservation status within the different biogeographic regions within their boundaries. As we have discussed, large carnivore populations function, and are best assessed, at supra-national scales. This implies that fragmenting the scale of assessment of a member state's portion of a wider populations to the different biogeographic regions within that member state is unlikely to convey any biologically meaningful information about the overall viability of large carnivores. This procedure will almost always **underestimate** the favourability of a species conservation state within a member state.

The procedure of aggregating species status across non-continuous sections of a biogeographic region (such as Alpine areas in the Alps, Apennines, Pyrenees, Scandes and Carpathians is even less biologically informative as individuals from these disjunct regions of Europe will rarely, if ever, interact and therefore cannot contribute to each others population viability. This procedure will almost always **overestimate** the extent to which a species conservation status is favourable.

Likewise, a focus on a priori defined biogeographic regions can **obscure** other important connections, or discontinuities, in the actual distribution of large carnivores which are critical for on-the-ground conservation (see section 11 for examples of the consequences of these issues).

It should be noted that in the Directive, biogeographic regions are only mentioned with respect to habitats types and the criteria for creating Natura 2000 sites, not for species conservation. Their connection to species reporting was an administrative decision taken when reporting guidelines were first developed.

In contrast, the biogeographic regions can serve as a **proxy** to document that the large carnivores have been allowed to spread and occupy a diversity of habitats and ecosystem types, which is essential to both conserve their "**ecological variation**" and promote the widespread connectivity which is essential for viability.

We would therefore recommend that assessment of conservation status at the biogeographic region within member state scale be focused on documenting the permanent **presence** of the species within a non-trivial proportion of that regions area within the member state in order to assess progress to restoration of ecological processes.

Although we have not considered it explicitly in this document, it would be possible to conceptualise a specific interpretation of what FCS means at the biogeographic level, i.e. an FCS_{BIO}, because it is clear that the biogeographic regions require a different understanding of the concept than that for the population and member state levels.

7.4 The special case of the golden jackal

Golden jackals represent a very special case for both FRP and FRR. In the last decade there has been an incredible expansion of the species both within and outside its historical “core” in southeastern Europe. Individuals have turned up in almost all European countries (except Portugal, Sweden, Luxembourg and Belgium) during recent years, including the boreal and arctic areas of Norway, Finland and Russia. Currently, the northernmost reproductions are in Estonia, Germany and Poland (Kaczensky et al. 2024, Mannil & Ranc 2022). The reasons for the rapid expansion are unknown (Krofel et al. 2007, Cinze & Klimpel 2022). Because this expansion is taking the jackal beyond the traditional areas that it has occupied for centuries it is impossible to predict if these colonising individuals will establish or disappear. This makes it hard for these newly colonised countries to set FRVs for the species, so that we would suggest that the only appropriate option at present is to set both FRP and FRR with an operator of “>CV” or simply as “unknown”.

For the countries in southeast Europe where they have had a stable presence over longer time scales the problem is the general lack of high quality / high resolution data on jackal densities and distributions. The species has never been subject to the same intensity of ecological study or monitoring as the four larger carnivores. This lack of data makes it hard to set numeric values for either FRP or FRR. For the smaller populations on the island of Samos and on the Peloponnese peninsula there is an urgent need for more intensive census work at the scale of the entirety of the populations to better understand the size of the populations and their distribution. It is unknown if Samos has any scope for population expansion so the only possible option is to say that FRP_{POP} and FRR_{POP} are equal to or greater than today (>=CV). The Peloponnese population clearly has scope for expansion but is hard to predict so setting FRR_{POP} and FRR_{POP} at greater than today (>CV) or unknown would seem justified.

Although the continental population is much larger it is subject to poorly regulated hunting in many countries. There is a need to establish both a system of reference areas that can be used to monitor changes in density over time, and establish a system of systematic record keeping that can help with detecting jackal expansion into new areas around the edges of the current distribution, and even detect contractions. Until these needed data are in place it is hard to suggest anything other than FRP and FRR at both member state and population scales should be set as equal to the current value (FRP = CV, FRR = CV).

7.5 Monitoring

Monitoring is an essential obligation under the Habitats Directive. Article 17 requires surveillance and reporting every 6 years. Article 18 mandates the necessary research (Louette et al. 2015). The fundamental definitions of Favourable Conservation Status in Article 1(i) implicitly require an assessment of trends in the amount of potential range and the quality of habitat within this range, as well as explicitly requiring data on the population dynamics of the species. It should also be noted that much of this data is needed for CBD indicator reporting under the GBF.

The suggestions for approaches to define FRP and FRR above will depend on monitoring data to determine the trajectory of the portions of the population within each member state and of the populations as a whole. The basics will include;

- Census of large carnivore population size that can be used to estimate the number of mature individuals (see Box 2).

- Indirect indicators of population trends in abundance.

- Distribution mapping in a manner that permits the separation of areas of reproduction / permanent presence and regions of occasional presence or vagrancy.

- Mapping of the permeability of the range – including areas of permanent distribution and connectivity corridors between them. At the very least this should focus on large infrastructure development, especially linear features that may obstruct connectivity.

Furthermore, because of the proposal to benchmark values on effective population size it would be highly useful, if not essential, to also monitor the genetical structure of the population including;

- Direct assessment of effective population size.

- Monitoring the allelic diversity.

- Documenting inter-population movements and geneflow.

- The minimisation / exclusion of including wolf-dog hybrids from estimates of wolf numbers.

Monitoring inter-population movements may be challenging in large populations because of the low probability of detecting immigrants. In such cases documenting a continuous distribution of reproductive units may be an acceptable, although inferior, surrogate. It should be noted that with less robust data there is a need to adopt a greater degree of precaution (see section 7.10).

The recent ruling by the CJEU in the case of wolves in northern Spain (C-436/22) also underline the need for surveillance at large scales, including transboundary scales, such that the impacts of management actions can be assessed. Coordinated and harmonised management across borders (both intra-national and inter-national) is viewed as an essential measure in large scale conservation planning and will be essential for the operationalisation of these guidelines in light of the fact that most large carnivore populations are transboundary in nature. Avoiding double counting, detecting migrants, and having comparable methods to fairly divide responsibility are just some of the reasons why a high degree of cross-border coordination is needed. Technical cooperation with respect to monitoring across borders is becoming increasingly common in Europe and there are many good examples of best practice. However, there are still examples of where this cooperation is sub-optimal and where there is a need to encourage better cooperation.

7.6 Threat assessments

The definition of FCS in the Directive invokes clear conditions on future prospects as well as present condition. To assess this the current reporting procedures involve large lists of pressures and threats (available online on EIONET's Central Data Repository https://cdr.eionet.europa.eu/help/habitats_art17) While these are exhaustive with respect to many issues related to habitat and environmental conditions and infrastructure related threats, they fail to recognise many of the threats that actually face large carnivores, or if they broadly cover the threat they do not identify the appropriate mechanism. For example, the threats posed by livestock husbandry (Threat category PA07, 08, 10) are not related to the livestock production per se, but to the extent to which appropriate husbandry methods are used to protect the livestock. The threats posed by forestry (Threat categories PB) don't deal with issues related to disturbance (e.g. of dens) or ensuring the supply of food (for example masting trees or berries for bears) or related to wildlife management of natural prey of large carnivores. Category PG08 does refer to hunting directly and on prey, but wildlife management is often determined in part by large herbivore damage to forestry, such that it is important that the interface between these two considerations, hunting and forestry, also considers the herbivores as a prey base for carnivores. The section on military action (Threat category PH) does not include border security fencing. Furthermore, there are no recognised threats related to poor institutional arrangements or lack of social acceptance related to conflicts between large carnivores and humans, or between different groups of humans over the way to manage large carnivores (Linnell 2013). One consequence of these threats may operate via illegal killing (which is recognised, threat PG11). 7.7 The process of population level assessment

Although not a legal requirement, in the 2008 guidelines it was hoped that member states would take the opportunity to form binding transboundary management plans that would coordinate management of the different parts of the populations that lie within different national borders. In the 16 years since their release no countries have yet made this step. While we would continue to encourage member states to make these plans it is apparent that other mechanisms need to be developed to allow population level assessments to be made based on the national level data submitted by member states. The European Environmental Agency already conduct post-submission aggregated analyses of FCS at the level of the biogeographic regions, so it would be possible to conduct a similar analysis for the large carnivore populations based on what is submitted.

7.8 The need for landscape scale planning

Because of this strong focus on connectivity it will become increasingly important to conduct population (or continental) scale landscape planning exercises to identify the critical areas for connectivity for all large carnivores (e.g. Hebblewhite et al. 2021, Oeser et al. 2023, Scharf & Fernandez 2018, Schnidrig et al. 2016, Vlkova et al. 2024). Plenty of data exists from telemetry and population monitoring studies to construct maps of suitable habitat. These layers should be examined for bottlenecks and barriers of impermeable infrastructure or other landuse developments that may need to be mitigated. Such mapping exercises will be important for (1) assessing realistic levels of connectivity under today's situation, (2) planning connectivity restoration exercises, (3) targeting conflict reduction measures, and (4) guiding future developments to prevent increased fragmentation.

This work could be reinforced if the Commission initiated regional population forums for knowledge exchange and assessment such as mandated by Article 18(1), and through landscape level planning (see section 7.7). It could be conducted in cooperation with regional initiatives – such as the Alpine and Carpathian Conventions – expanding on existing activities (e.g. Hacklander et al. 2021, Schnidrig et al. 2016). On a continental scale it could also be embedded in a pre-existing format of European Species Action Plans and involve ongoing activities like the EU Platform on Coexistence between People and Large Carnivores or expert groups such as the Large Carnivore Initiative for Europe. In order to ensure legitimacy and avoid conflicts over contested knowledge it would be best if such exercises were conducted using a broad consortia of technical experts and with a high degree of consultation with stakeholders and competent authorities in member states.

7.9 Consequences of the multi-scale approach

When splitting FRV into two different concepts at least two scales (we consider member state and population scales here, but the discussion on member state scale also needs to be downscaled to subnational units too – or be applied to biogeographic regions) the question arises if a member state can reach FCS even if the overall population has not, and vice versa, can a population be viewed as being at FCS even if all contributing member states have not yet reached their objectives? In other words there is a question about the relationships between FCS_{MS} and FCS_{POP}.

It is certainly possible for a member state to reach its FRV_{MS} even if its neighbours have not yet reached theirs and if the overall population is not at its FRV_{POP} values. This should be acknowledged, although if this should extend to using the term FCS at the member state level or not is unclear. On one hand, member states can only be held responsible for their own actions within their own borders. On the other hand, for some smaller countries at least, these FRV_{MS} may be so small that there is little overall viability of the species concerned. Because the Directive definitions of FCS focus on absolute outcomes (long-term conservation) it would be logical to withhold the status of FCS at a member state level until the population as a whole can be judged to have reached it, although it should be acknowledged then that the member state in question has delivered on its obligations even if the sum of the neighbours' actions have not.

However, the opposite situations are not necessarily true. A member state cannot claim to be at FCS just because the overall population is at FCS without making an own contribution and reaching its FRVs (rejected by CJEU Case C-601/22). Furthermore, if enough member states have contributed enough to bring the overall population to a level compatible with its FRV_{POP} it is logical that the population as whole can be declared at FCS (FCS_{POP}), as well as the member states (FCS_{MS}) that have fulfilled their obligations, even if one or more other member states have not yet reached their goals.

It is important to note that the recent ruling by the CJEU in case C-436/22 underlines that it is highly problematic if the administration in one unit begin killing carnivores if the wider population has not yet reached FCS. The implication is that even if a country has reached FCS_{MS} it needs to be extremely restrictive with management until the overall FCS_{POP} has been reached. IT is important to note that this also concerns cases of Annex V as well as Annex IV designation.

7.10 Precautionary considerations

The precautionary principle is enshrined in Article 191(2) of the Treaty on Functioning of the European Union "*Union policy on the environment shall aim at a high level of protection taking into account the diversity of situations in the various regions of the Union. It shall be based on the precautionary principle ….*".

This has consequences for all aspects of these guidelines. Operationalising these guidelines requires conducting many analyses where the outcome is dependent on both empirical data and a set of assumptions about many ecological and genetical properties of species' populations. Furthermore, all calculations in science require making decisions about acceptable probabilities of certain outcomes happening. There is considerable variation in the degree of scientific knowledge about European large carnivore populations. A logical application of the precautionary principle is that where up-to-date and contextual knowledge lacks there will be a greater need to apply precaution. As a result, many of the rules-of-thumb and heuristics that we apply include a high degree of precaution. If more specific data or analyses are available to circumvent these heuristics there may accordingly need to be less precaution. Key parameters where precaution may be considered concern (1) the heuristic of one effective migrant per generation which is minimal, and (2) the 50:500 rule, where 100:500 or even 100:1000 have both been proposed as more cautious rules-of-thumb.

8 Summary of proposal for FRVs at population and member state levels

This section presents the results of the background and discussions in all previous sections in a brief operational overview / checklist fashion.

For all levels there is a requirement that no favourable reference values can be lower than at the point in time when the member state entered the EU (with the possible exception of situations where local population densities were artificially high, for example because of supplementary feeding). It also follows that improvements in status in another member state with which a population is shared or in connectivity between populations cannot automatically be used to then justify downgrading (degrading) the level of ambition in FRVs.

For all scales both FRP and FRR should be considered together so that the value with the larger requirements takes precedence (i.e. if fulfilling FRR requires occupying more areas than would be strictly necessary to satisfy FRP when viewed in isolation then it is the FRR that sets the overall level for FRP, and vica versa).

There should obviously be no expectation for any administrative unit at any level to host more large carnivores than the habitat can support in a sustainable manner.

In countries with federal structure or other structures that delegate some management authority to subnational levels the principles of the member state level can be downscaled to suit their size, location and ecological preconditions, however it is essential that higher level coordination is maintained.

8.1 Population Level

FRP_{POP}:

- The sum of all member state contributions (plus non-member state contributions) represents an effective population size (N_e) greater than 500.

- For a unit to be assessed there must be either a continuous distribution (i.e. the current large carnivore populations) or sufficient exchange of individuals across areas of non-continuous distribution to ensure effective geneflow.

- Where there is a connection to a population in a country that is not bound by the Habitats Directive or Bern Convention there is a higher threshold to demonstrate the absence of barriers (border fences) and documentation of effective geneflow.

FRR_{POP}:

- Composed of a continuous and aligned area of interconnected distribution, or potential distribution, with sufficient habitat and without serious internal barriers to movement or discontinuities beyond average dispersal distances.

- An area large enough to embrace the FRP_{POP} with realistic population densities.

- Including potential connection corridors to neighbouring populations through which individuals can regularly disperse even if they are not resident.

8.2 Member State Level

FRP_{MS}:

- It is recommended that the minimum requirement is that FRP_{MS} should be large enough so that the sum of FRP_{MS} values of all member states (and non-member states included) within the shared population reaches FRP_{POP} (N_e>500).

- With this overall goal the FRP_{MS} goal could be of a size that represents a fair and proportional sharing of the common obligation between member states towards reaching the overall FRP_{POP} while taking into account the combination of ecological, social, cultural and economic requirements and regional and local characteristics of the different member states (see Box 3 for a very rough approximation of some key issues).

  - For countries with "very small" (i.e. less than 10.000 km^2) areas of potential habitat the contribution should allow for the permanent presence of reproductive units of large carnivores in a significant proportion of the country, but without setting quantitative goals.

  - For countries with "small to medium" (i.e. from 10.000 km^2 to 50.000 km^2) amounts of potential habitat the contribution should be scaled to quantitative goals benchmarked against either MVP or an effective population size of 50 (or 100 if applying a large degree of precaution).

  - For countries with "large" (i.e. > 50.000 km^2) amounts of potential habitat the contribution should be scaled far above an N_e of 50, ideally as close to N_e of 500 as possible.

- To satisfy the ecological functionality conditions there will also need to be an actual permanent presence of reproducing units of the species in the Natura 2000 sites designated for the species in question, biogeographic regions and range of ecological conditions defined within the FRR_{MS}.

Note: A special case concerns countries that host portions of multiple populations. For very-small and small-to-medium sized countries it is reasonable that their FRPs refer to their total contribution. However, for large countries it will both reasonable and necessary that there is a need for a substantial contribution to each of the populations that they co-host, although the cumulative contribution they make to all populations within their territory should be considered when deciding on the fair level of sharing responsibility between countries.

FRR_{MS}:

- Composed of a continuous and aligned area of interconnected distribution, or potential distribution, with sufficient habitat quality and without serious internal barriers to movement or discontinuities beyond average dispersal distances.

- Habitat types of sufficient quantity and quality are included within the range.

- Aligned with FRR_{MS} of neighbouring states with which the population is shared to ensure connectivity between member states and satisfy the requirements for FRR_{POP}.

- Overlapping all Natura 2000 sites designated for the species.

- Overlapping all biogeographic regions within the country that can be considered natural range.

- Overlapping all relevant ecological conditions, ecosystems and prey communities.

9 Implementing these new guidelines

9.1 Check lists for assessing Favourable Conservation Status based on new FRVs

Based on the argumentation presented in section 7, and summarised in section 8, together with the basic principles of key concepts outlined in section 1 this section summarises the key elements of FRVs and their linkage to FCS into a “checklist”. It is important to bear in mind the multiple caveats and special cases identified in the previous sections, and additional elements of FCS classification that are used during the formal Article 17 reporting process. The precautionary principle should also be exercised with respect to data quality and uncertainty in parameter estimates.

Population level

	Parameter	**Yes/No**
FRP_{POP}		
1	Does the sum of all member state contributions represents an **effective population size** greater than **500**?	
2	Is the population **trend** positive or stable?	
FRR_{POP}		
3	Is the FRR composed of a continuous and aligned area of **interconnected distribution**, or potential distribution?	
5	Is the range stable or increasing?	
5	Is there **sufficient habitat** and without serious **internal barriers** to movement or **discontinuities** beyond average dispersal distances?	
6	Is the **area large enough** to embrace the FRP_{POP} with realistic population densities?	
7	Is the **prognosis for the habitat** quality and connectivity positive?	
8	Are there **potential** connection corridors to **neighbouring populations** through which individuals can regularly disperse even if they are not resident?	
9	Have all **genetically distinct units** or subspecies been included in the range?	
FCS_{POP}	If answer to **all** parameters (1-9) is yes – then FCS has potentially been achieved – if the answer to any parameter is no, then FCS_{POP} has not been achieved.	

Member state level (potentially down-scaleable to sub-national levels)

	Parameter	Yes/No
FRP_{MS}		
1	Is your FRP greater than or equal to when you **entered the European Union**?	
2	Is your FRP large enough so that when **summed** with FRPs from other member states (and other contributing states) sharing a population it will allow an effective population size of at least 500?	
3	Does your population have a stable or positive **trend**?	
4	"Very small countries" – do you have the permanent presence of reproductive units of large carnivores in a significant proportion of the country?	
5	"Small to medium size countries" – is your population size greater than a demographic MVP or an effective population size greater than 50?	
6	"Large countries" – is your FRP for each segment of a population that you co-host much greater (proportional to the available habitat) than an effective population size of 50?	
7	Are reproducing units of the species present in the full range of Natura 2000 sites, biogeographic regions and relevant ecological conditions?	
FRR_{MS}		
8	Is the FRR composed of a continuous and aligned area of **interconnected distribution**, or potential distribution?	
9	Is the range **stable or increasing**?	
10	Is there **sufficient habitat** and without serious **internal barriers** to movement or **discontinuities** beyond average dispersal distances?	
11	Is the **area large enough** to embrace the FRP_{POP} with realistic population densities?	
12	Is the **prognosis for the habitat** quality and connectivity positive?	
13	Is the FRR **aligned** with neighbouring states to ensure sufficient connectivity to allow FCS_{POP} to be attained?	
14	Does the FRR overlap all **Natura 2000** sites designated for the species?	
15	Does the FRR overlap all **biogeographic regions** within the country that be considered natural range?	
16	Does the FRR allow for the presence of the species in all **ecological conditions**, ecosystems and prey communities?	
17	Are all subspecies of distinct genetic populations included?	
FCS_{MS}	If the answer to **all** parameters 1-3 and 7-17 and **either** parameter 4, 5 or 6 (depending on your situation) is yes – then FCS has potentially been achieved – if the answer to any of these parameter is no, then FCS_{MS} has not been achieved.	

9.2 Preparatory actions need for implementing the new guidelines

Implementing these new guidelines is not a trivial task. There is a need for many scientific, administrative and possibly even political actions. It is therefore unlikely that member states will be able to fully operationalise them for the current reporting cycle. The following represents a list of some of the most basic actions that are needed. It should be noted that for many parts of Europe most of these elements already exists in one form or another.

Recommendations for scientific and technical actions

- Analysis of data to develop **conversion factors** between monitoring metrics and numbers of mature individuals.

- Analysis of data to develop best estimates of the **N_e / N_c ratio** for all the large carnivore species in their different contexts.

- Assessment of degree of **connectivity** between existing “populations” to produce **revised units of assessment** (possibly merging some populations) relevant for calculation of effective population size.

- Development of comparable metrics for **population level assessment** of transboundary populations using harmonised methodology.

- Assess **current effective population size** for this population and identify **population level FRVs**.

- Development of new, and integration of existing, **habitat suitability maps** as well as maps of potential **connectivity** and identification of **barriers**.

- Make broad assessments of approximate **ecological carrying capacities** of these distribution areas under different scenarios.

- Examine **overlap** with Natura 2000 sites, biogeographic regions and major ecosystem types within the member state’s area.

- Identification of **social, economic and cultural** considerations that may need to be considered.

- Decide on **member state level FRVs** needed to ensure that population level FRVs can be reached or maintained while ensuring a fair and proportional distribution of the responsibility among member states.

Recommendations for administrative actions

- Adjustments to Habitats Directive **reporting forms** to better accommodate transboundary reporting and the metrics described in these guidelines.

- Establishment of a **working group** to make post hoc transboundary assessments of population status based on reported or published data in situations where member states do not deliver coordinated reporting of shared populations. This group could make a first assessment of the situation based on the current round of reporting to provide better practical guidance for its application in the next reporting cycle.

- Creation and facilitation of **transboundary forums** for discussions around policy coordination, and if possible, the creation of transboundary management plans.

- Develop a step-by-step **user guide** for implementation in the context of the next reporting cycle.

9.3 Subjectivity, scientific uncertainty and scope for member state discretion

Although our rational has been to align the best scientific practice with legal and policy framings it is clear that the approaches we outline can we addressed in different ways. It was never our intention to develop a prescriptive cookbook. For example, multiple approaches exist for calculating effective population size and minimum viable populations or for modelling habitat suitability. Similarly, many of these models require placing values on parameters for which there may be no empirical basis, or on transferring values from different study populations to deal with data gaps (see section 12). In some cases there will be a need to make predictions about future or potential conditions. Finally, most modelling and analytical approaches involve making choices about acceptable probabilities of risk of different outcomes and different ways of dealing with uncertainty.

The decisions made on these matters are likely to influence the outcome of the calculations. To a large degree this is inevitable as it reflects a diversity of scientific approaches and the constant development of methods and data availability in the field. There is not a single right way of doing this type of science, although there are many wrong ways. This also opens for a certain degree of member state discretion with respect to how they go about setting their FRV reflecting different thresholds of risk acceptance.

Because of the controversial nature of large carnivore management we strongly recommend that all processes are conducted transparently and that all data, calculations and models are made available for critical assessment by scientific peers and colleagues.

C: Scenarios: setting Favourable Reference Values under different parameters

These sections are intended to illustrate the real-world consequences of different criteria included in our proposal for FRVs.

10 Natura 2000 coverage

These maps illustrate the extent to which Natura 2000 sites designated for large carnivores are already included within the distribution range of the species for which they were designated, thereby also identifying sites designated for that species which are not yet included within the current distribution range of the species. Several member states took out an exception for some of the large carnivore species with respect to Annex II. For wolf, an exception was taken by Estonia, Latvia, Lithuania, Finland and Spain north of the river Duero. For bears, an exception was taken by Estonia, Finland and Sweden. For lynx, an exception was taken by Estonia, Latvia and Finland. Golden jackals are not listed on Annex II. Swedish files have been submitted to the EU but could not be downloaded and could therefore not be included. For wolves, bears and lynx we first show an overview of continental Europe and then zoom on a selection of additional areas for illustration. The current distributions of large carnivores are taken from Kaczensky et al. (2024), including all categories of distribution (permanent, occasional and unclassified).

In the case of wolves (Figure 2), the upper left map illustrates the fact that most of the Natura 2000 sites created for wolves are currently covered by present day wolf distributions. The upper right figure shows a major exception in the case of Iberia where many Natura 2000 sites in the south and southwest of the country are currently outside of wolf distribution. At the time that Spain entered the EU the Sierra Morena wolf population was still extant, but it is now viewed as being extirpated since the 2000's. Wolves disappeared in Extremadura earlier, in the late 1980's or 1990's. Spain also has many Natura 2000 site sin the Pyrenees that may soon be occupied if expansion from both the west and the north continues, potentially providing a major stepping stone linking the two populations. The bottom left figure shows that the Alpine and Dinaric-Balkan wolf populations have good coverage of the designated Natura 2000 sites, with the exception of one in northern Italy (in the region of Parco Regionale delle Orobie Bergamasche).

The current distribution of lynx in central Europe is clustered around sites designated for lynx (Figure 3). However, there are multiple sites for the species that are currently unoccupied, especially in the southern French Alps, the eastern Alps of Italy and central Austria. The latter two regions represent vital connections that need to be established to reconnect populations and will therefore have a vital role to play in the future.

The examples selected for bears (Figure 4) show contrasting situations. The left hand map shows how the Carpathian bears have a very good coverage of the designated sites, with the exception of an area in southwest Romania (centered around Parcul National Semenic-Cheile and Parcul National Cheile Nerei - Beusnita) which could be important for connections towards Serbia. The right hand maps focuses on the connection area between the Alpine and Dinaric-Balkan-Pindos populations. Most striking is the area in central Austria where many sites were created for the small bear population that was extant at the time but which has now disappeared. There are also unoccupied sites in Italy that could help establish connection between the Alps in Italy and Slovenia, and provide space for the population in the Italian Alps to expand westwards.

In addition to the sites designated for the species the maps show that there are many other Natura 2000 sites in these connection zones that were not designated for large carnivores at the time, but which could become important to foster connections between these expanding populations. The maps also show the very large disparity between different countries in the size and configuration of their Natura 2000 networks. However, in no case is the Natura 2000 network enough to ensure conservation or connectivity of large carnivores without their presence in the surrounding landscape.

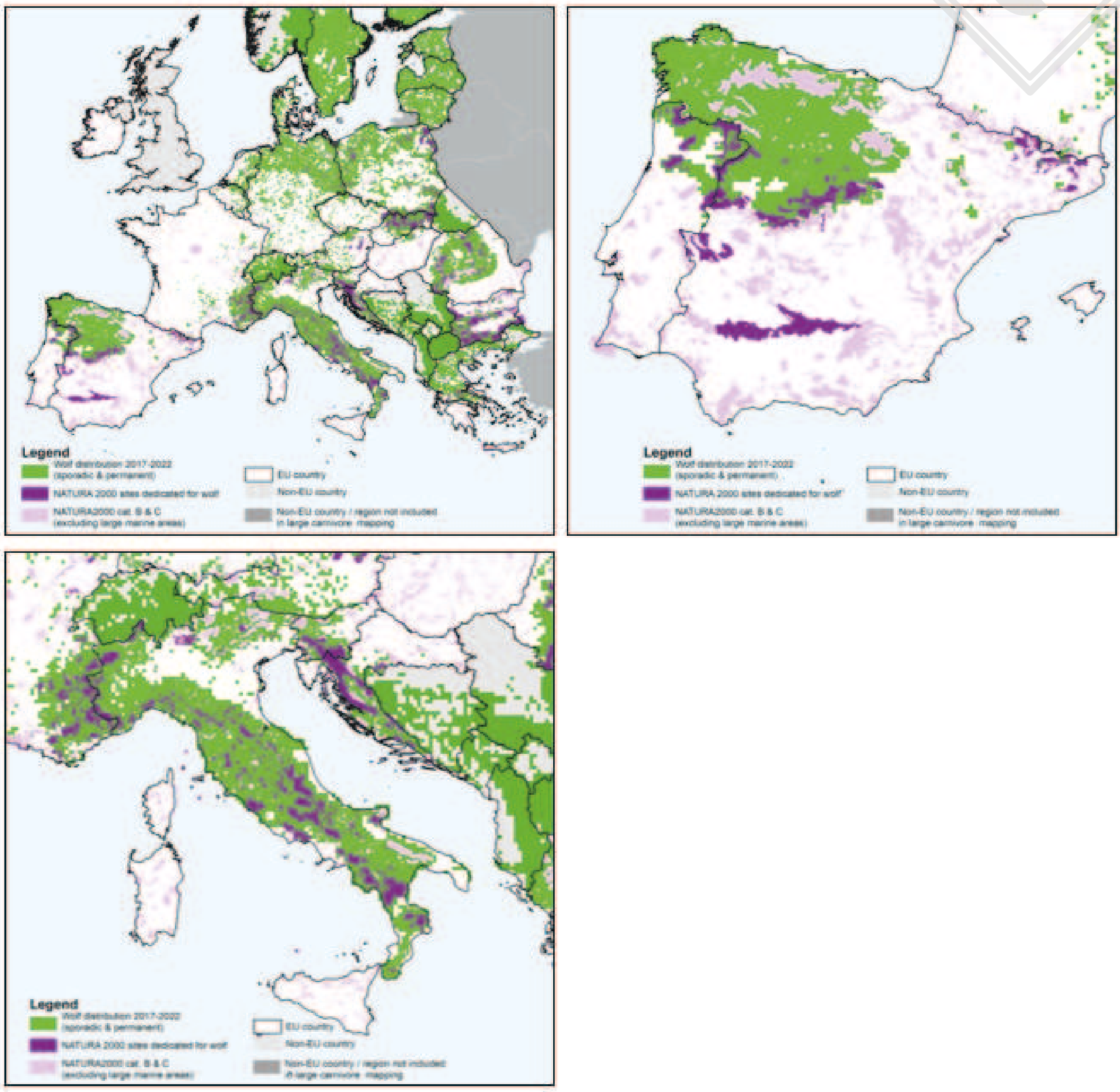


Figure 2. Overlays between current wolf distribution and Natura 2000 areas designated for wolves.

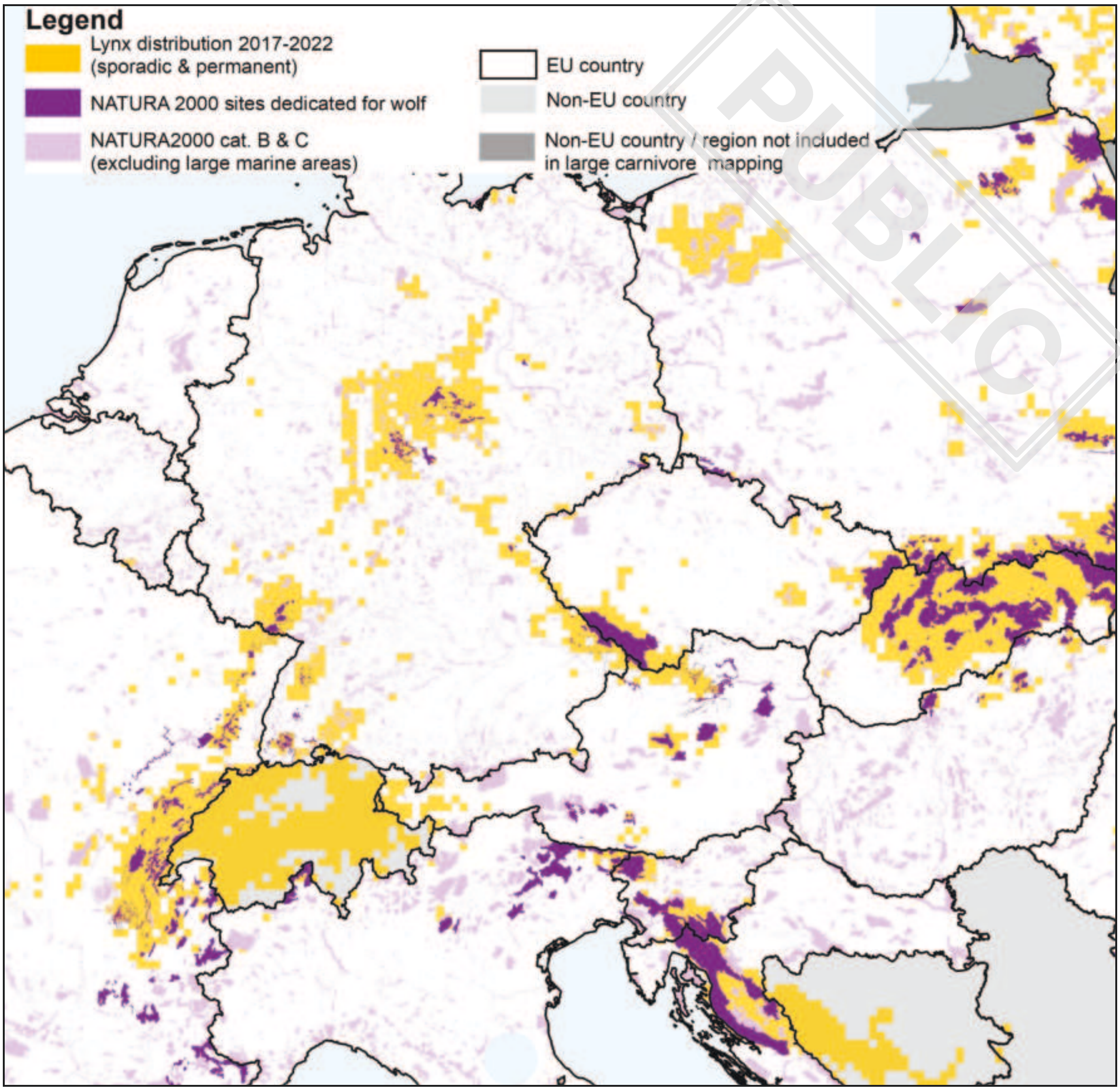


Figure 3. Overlays between current lynx distribution and Natura 2000 areas designated for lynx.

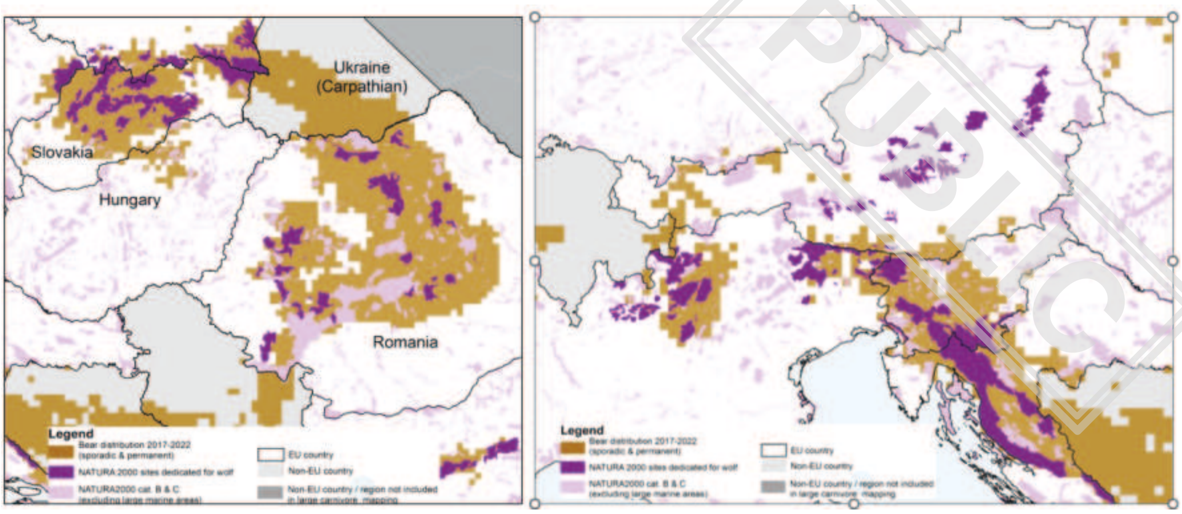


Figure 4. Overlays between current bear distribution and Natura 2000 areas designated for bears.

11 Biogeographic region coverage

These maps (Figure 5) overlay current large carnivore distribution with the biogeographic regions to illustrate the extent to which large carnivores are currently found in the regions. Overall one of more species is present in all of the biogeographic regions, but the extent to which they are covered varies. The Alpine, Boreal and Continental regions have by far the highest degree of coverage, while the Atlantic and Pannonian Basin have the least.

If having at least some presence in all regions is a requirement to satisfy FRR at both member state and population scales gains will be needed in the Atlantic biogeographic region. The recent expansion of wolves into the Benelux countries and northeast Germany provides good coverage, as does the long term presence of wolves in northeastern Iberia. However, France is the country with by far the largest area of Atlantic and as of yet has almost no wolf presence. Fostering modest expansion in the northeast (connecting Alpine and Central Europe wolf populations) and the south (connecting Alpine and Iberian wolf populations) would satisfy twin goals of building population connections and bringing wolves into this under represented biogeographic region. From the perspective of habitat this expansion would seem to be technically feasible.

For bears it is very unclear how much scope there is for further expansion in the Atlantic region outside of the northern part of Spain (Cantabria) and southwest France (Pyrenees). The degree of human landscape dominance in most of the region in other parts of France and the Benelux would seem to represent irreversible changes from the perspective of bears. Bears are also only present on the fringes of the Continental region at the ecotones with the Alpine. It is unclear how much scope there is for bear expansion into the Continental because of irreversible habitat changes from a bear's perspective. At best it'll be a question of expanding how far outside the Alpine regions bears are able to expand.

Lynx are also barely represented in the Atlantic region. Realistically speaking, the area where there is a chance of improvement on short-term time-scales is in northwestern Germany and the Benelux countries. This would have the advantage of allowing expansion and enhancing connections between many of the small lynx populations and occurrences that are scattered across the region. Lynx also only have a small presence in the Mediterranean region on the coast of Slovenia and Croatia. There is some scope for expansion in Croatia, although Greece holds the greatest potential. Unfortunately there is still no confirmed presence of the Balkan lynx there.

The Pannonian basin stands out as a biogeographic region with very little large carnivore presence (apart from golden jackals), although lynx, wolves and bears are present all around it. From a habitat perspective it is unlikely that it represents suitable habitat for bears. It is also marginal for lynx. In contrast, there should be potential for wolves to colonise at least parts of it. However, the area needs a detailed habitat analysis. There are abundant sources of colonising carnivores to the north and east (Carpathians), but connection to the south and southwest is severely hindered by Hungary's border security fence on the Serbian and Croatian borders.

Overall, focusing on the FRR requirement and fostering at least some large carnivore presence in all suitable biogeographic region could be achieved with relatively modest expansions of existing populations with the greatest need involving the Atlantic and Pannonian basins.

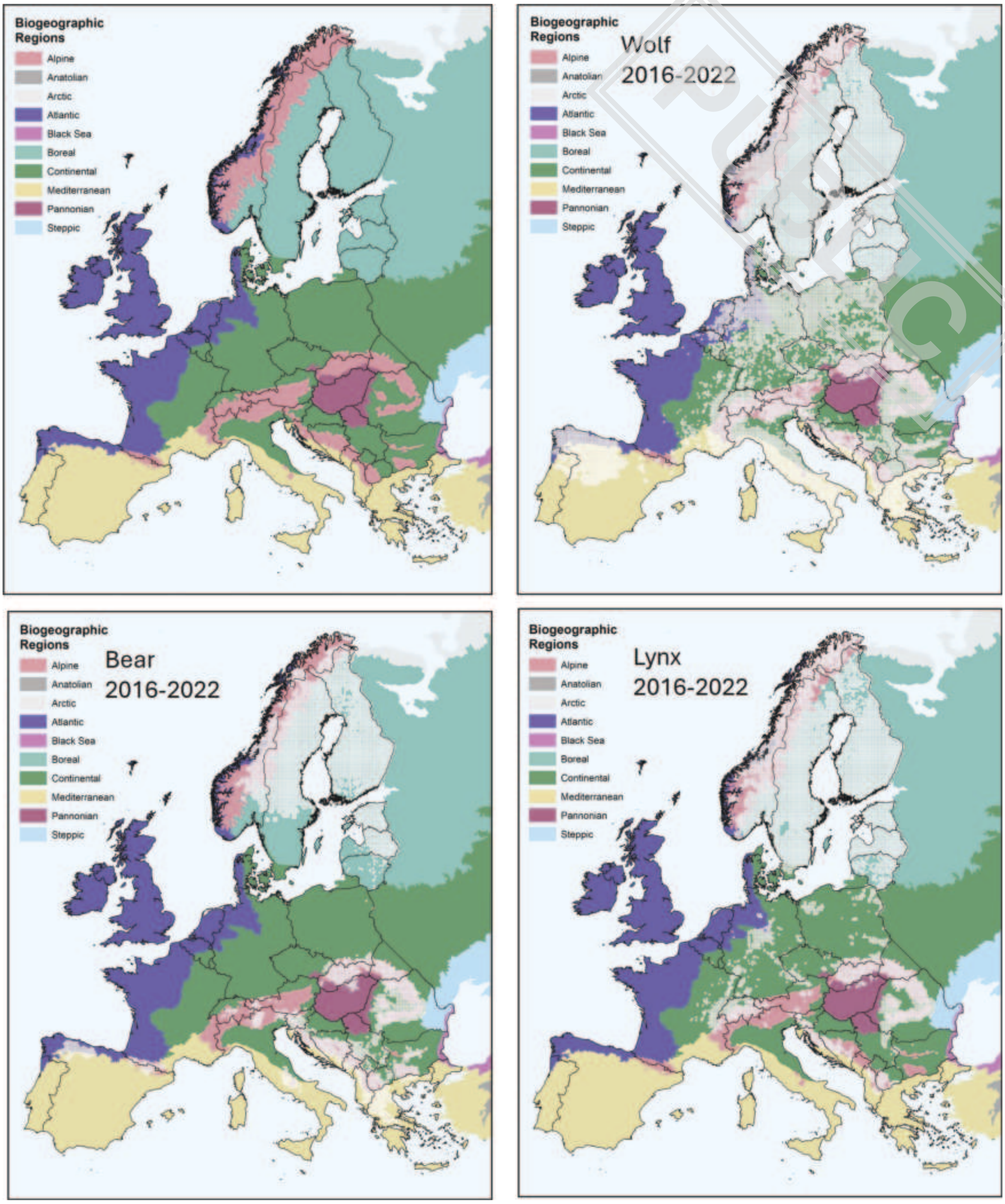


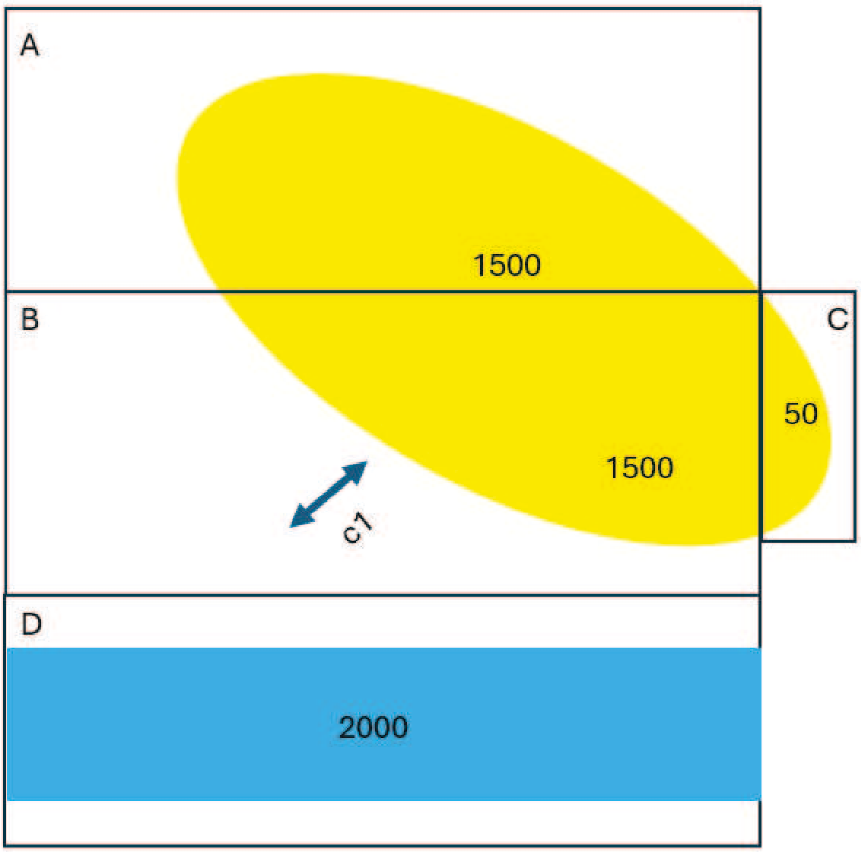
Figure 5. Overlays between wolf, bear and lynx distributions (data from 2016 to 2022, all distribution types combined, Kaczensky et al. 2024) and the biogeographic regions recognised by the Habitats Directive.

12 Population size and distribution with respect to proposals for FRVs: model scenarios

In this section we provide some **illustrative model scenarios** to illustrate how different population situations might translate into different FRV assessments. Rather than basing the scenarios on real life assessments of named populations we have chosen to show some model or idealised situations that loosely reflect real life situations. Although Kaczensky et al. (2024) presents updated distribution maps and status assessments of populations there are several parameters that are essential for accurate assessment that are not presented in that report. This includes the assessments of actual connectivity between regions and accurate estimates of parameters linking population size estimates with effective population size (i.e. the N_e / N_c ratio). The use of idealised models allows the importance of this uncertainty to be revealed and prevents a too hasty preliminary assessment of a real life population / national situation without all of the necessary information becoming available. It should therefore be viewed more as an illustration of what our proposal for FRPs **could** mean in real life, more than an assessment of how different member states are **actually** performing.

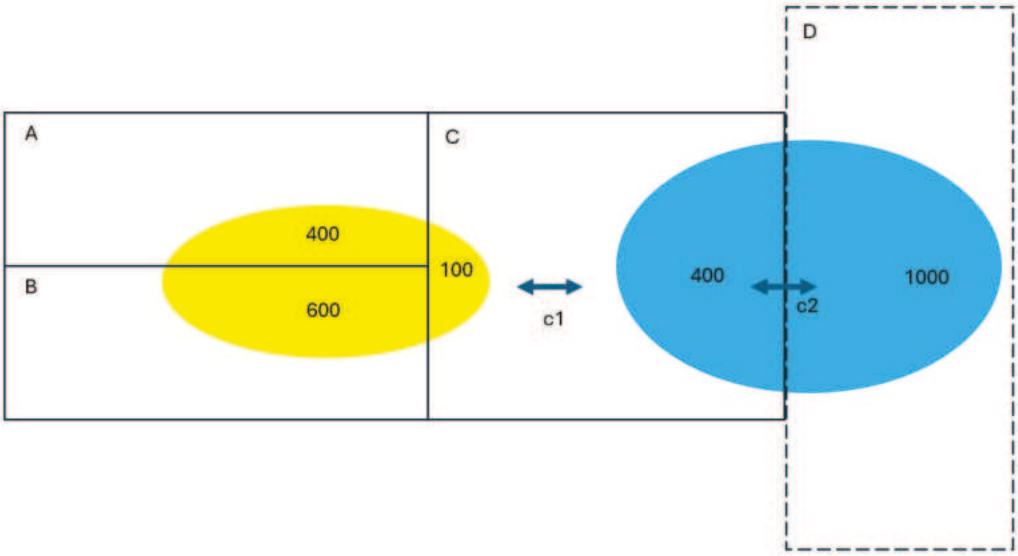
Example 1. A simple scenario where two large countries (A, B) and a small country C share a common population (Yellow) under three different N_e/N_c ratios and with different degrees of connectivity to another population (Blue). The outcomes in green show cases where FRPPOP and FRP_{MS} are above their expected thresholds (N_e >500 at population level, and N_e >50 in each of the large countries and country C has made a significant contribution with respect to its potential. Outcomes in yellow show cases where each of the countries have made significant contributions, but are not enough to bring the population as a whole to a level where the N_e>500. All population sizes refer to number of mature individuals.

N_e/N_c	Country A	Country B	Country C	Population
	1500	1500	50	3050
No connectivity between Blue and Yellow populations				
0.1	150	150	5	305
0.2	300	300	10	610
0.3	450	450	15	915
Enough connectivity between Yellow and Blue populations so that they can be considered a single genetic unit				
0.1	150	150	5	505
0.2	300	300	10	1010
0.3	450	450	15	1515



Example 2. This illustrates 3 large countries (A-C) that are all signatories of a binding treaty, bordering onto a fourth large country (D) which is a signatory – but where there is a real connection. We consider scenarios with different N_e/N_c ratios (0.1, 0.2, 0.3), different degrees of connectivity between the Yellow and Blue populations (c1 and c2), and situations where the contribution from country D can be considered or not. The outcomes in green reveal where the overall populations can be considered to have a large enough population to reach a FRV_{POP} that would be acceptable as FCS (N_e>500), and where the national FRPMS are at a level where N_e>50 and the overall population is at N_e>500. Outcomes in yellow illustrate situations where each country has made a meaningful contribution by reaching a level where N_e>50, but where the overall populations have not yet reached a combined level of N_e>500. For these cases to turn green would require increasing their population sizes. All population sizes refer to number of mature individuals.

N_e/N_c	Country A - Yellow	Country B - Yellow	Country C - Yellow	Country C - Blue	Population – Yellow	Population Blue
	400	600	100	400	1100	400 (1400)
No connectivity between Yellow and Blue population – and unable to consider portion of Blue population in country D						
0.1	40	60	10	40	110	40
0.2	80	120	20	80	220	80
0.3	120	180	30	120	330	120
No connectivity between Yellow and Blue population – but able to consider portion of Blue population in country D						
0.1	40	60	10	40	110	140
0.2	80	120	20	80	220	280
0.3	120	180	30	120	330	420
Enough connectivity between Yellow and Blue population to consider as one genetic unit – but unable to consider portion of Blue population in country D						
0.1	40	60	10	40	150	40
0.2	80	120	20	80	300	80
0.3	120	180	30	120	450	120
Enough connectivity between Yellow and Blue population to consider as one genetic unit – and able to consider portion of Blue population in country D						
0.1	40	60	10	40	250	250
0.2	80	120	20	80	500	500
0.3	120	180	30	120	750	750



Appendix 1 Current practices associated with setting FRVs in a selection of European countries

Member State reports from the 2013-2018 reporting cycle

The overview of member state reporting of Favourable Reference Values from the 2013-2018 reporting cycle indicates that very few have chosen to give numerical values. Instead most have chosen to give operators such as “more than”, “less than”, or “approximately equal to” (Table 3). With the exception of Sweden which has high quality monitoring systems in place, most of the countries that have given numerical values only have relatively poor monitoring systems in place that lack the precision to produce such exact numbers, or document where the current status is with respect to these values (Kaczensky et al. 2024). Paradoxically, many countries that have good monitoring and research systems in place have chosen to avoid giving numerical values. Based on these values it is impossible to determine the extent to which member states are achieving their goals or not. Many indicate that they are far below their FRVs, although a few indicate that they are currently above (Slovenia and Sweden for bears). It should be noted that all numbers in these tables refer to the total number of carnivores and not to the effective population sizes that form the basis of these new guidelines.

Table 3. Overview of Favourable Reference Values provided by member states in the 2013-2018 Article 17 reporting cycle on biogeographic level. For each country the numerical values have been summed for all biogeographic regions. Where different operators were used for different regions we have shown both symbols separated by a comma. “N/” and “x” indicate that no data was given. Source: extracted from https://nature-art17.eionet.europa.eu/article17/species/summary/

Country	Wolf		Bear		Lynx		Jackal		Wolverine	
	FRP	FRR	FRP	FRR	FRP	FRR	FRP	FRR	FRP	FRR
AT	N/	N/	>>	>>	>	> , >>	N/	N/		
BE	N/	N/								
BG	920	122000	525	28400	≈	21300	38126	123900		
CZ	>	>	N/	N/	>	>	x	x		
DE	>>	>>			>>	>>				
DK										
EE	≈	≈	≈	≈	>	≈				
ES	578 + x	≈ , >	492	≈						
FI	>	≈	≈	≈	≈	≈			≈ , >	≈
FR	≈ , < , N/	≈ , N/	>	>	> , > , x	> , ≈ , x				
GR	>	≈	>	≈	x	x	x	≈		
HR	>	≈ , x	≈	≈ , x	>> , x , x	> , ≈ , x	800	≈		
HU	>	>			>>	>	≈	≈		
IT	≈	≈	> , >>	≈ , >	>>	>>	≈	≈		
LV	300	64589	30	X	600	64589				
LT	≈	≈			>	≈				
LU	>>	>>								
NL	N/	N/			N/	N/				
PL	≈ , >	≈ , >	≈	≈	>	> , ≈				
PT	≈ , >	≈ , >								
RO	2700	≈	5960	≈	2300	≈	1900 , x	≈		
SI	≈	≈	<	≈	>>	>	≈	≈		
SK	≈ , >	≈ , >	≈	≈	> , ≈	>				
SE	300 , N/	238800 , N/	1400	315900 , N/	870 , N/	421400 , N/			600	293700

Detailed examination of some recent FRV processes

This represents a summary of a selection of national level processes that have led to the formulation of concrete FRVs for various large carnivores (mainly wolves, but also bears, lynx and wolverines). It is not a complete overview, but is based on those that were most accessible in terms of language, most complete, and most clearly linked to official policy processes. There are other good expert reports (e.g. Hulva et al. undated, Jansman et al. 2021) and national plans that were not included because of time constraints or uncertainty about their official status with respect to policy. Furthermore, several countries are currently working on models at the time of writing but are not yet finished.

Lynx in the Bohemian-Bavarian-Austrian lynx population

Under the auspices of an Interreg funded project (3Lynx Project) the responsible management agencies of Bavaria, Upper Austria and the Czech Republic, together with a range of scientific and NGO partners have developed a conservation strategy for lynx in the shared 3-country population. They developed a shared vision "to restore and maintain, in co-existence with people, a viable lynx population within the Bohemian-Bavarian-Austrian border region connected with other metapopulations in Central Europe" which embraces the idea of a multi-scaled population approach (i.e. the three country population as a part of a wider meta-population).

The strategy explicitly links an interpretation of Favourable Conservation Status to the IUCN Red List Criterion D (of 1000 mature animals). Because this large number cannot fit into the available habitat they have pragmatically adopted a two-pronged strategy of aiming for 250 mature lynx in the three country border population while securing genetic exchange with neighbouring populations to secure a meta-population with at least 1000 mature individuals in total. This adjustment of objective from 1000 to 250 is in line with IUCN Red List practices if there is sufficient connectivity. The objective of at least 250 mature animals is further specified as 165 reproducing females and 85 mature males. The strategy also identifies a large number of actions to monitor demographic and genetic aspects of the population, improve landscape permeability for connectivity, and evaluate the potential for assisted (translocation) connectivity if needed. There is, however, little discussion of the details of the wider meta-population connectivity. New continental scale maps of lynx habitat are now emerging (Iannella et al. 2024, Oeser et al. 2023) as are guidelines on connectivity analysis (Potocnik et al. 2024) which will support such larger scale planning.

Source: Czech Ministry of Environment (2020).

Brown bears, wolves and lynx in Estonia

Estonia has had a series of management plans for its large carnivores since at least 2002 which are updated every 10 years. The current plan is for the period 2022-2031 and focuses on the large carnivores present in Estonia while making reference to the wider shared Baltic population and its neighbor, Russia. The current plan states that the goal for Estonia is to ensure demographically viable populations (extinction risk <5% in 100 years) within Estonia. Threshold values for favourable conservation status are set for each species, formulated in terms of numbers of reproductions (the monitoring unit). These are 20-30 wolf packs, >80 lynx reproductions, and >70 brown bear reproductions. All values are expressed as being before any annual harvest (i.e. late summer / autumn values). Additional targets are formulated based on the total number of individuals after any annual harvest (i.e. spring); >140 wolves, >350 lynx, >650 bears. These targets are based on (1) must exceed the minimum of demographic viability within Estonia, (2) a need to reach Baltic wide populations of 1000 breeding individuals, and (3) the share of habitat and prey status within Estonia compared to other Baltic countries.

Source: Anonymous (2022).

Wolves in Finland

An extensive expert report (Mäntyniemi et al. 2022) provides an exhaustive conceptual and empirical analysis of FCS and the Finnish wolf situation. It is built on the premise that FCS, and associated FRVs, allows for decision makers to make certain choices about subjective parameters, although it must of course be within the frames of lower thresholds (set by a Minimum Viable Population and the size of the population when they entered the EU) and upper limits (set by the ecological carrying capacity). The report directly links FCS to various viability concepts; demographic viability, short-term genetic viability and long-term genetic viability. The issue as to whether Finland needs to opt for short-term or long-term genetic viability depends on the extent to which it can consider the connectivity with the larger Russian populations or not, however, it is stated that long-term genetic viability cannot be achieved within Finland's borders. By extension the report then focuses on maintaining demographic and short-term genetic viability as the goals for Finland with respect to FCS. Empirical data from field studies – reproduction and survival of collared wolves, population monitoring data and genetical data – was used to parameterize a set of viability models. The availability of >20 years of combined genetical and ecological monitoring data has permitted the calculation of population specific estimates of the Ne/Nc ratios (ranging from 0.3 to 0.5) and inbreeding rates.

The report is based on conservative choices, namely adopting a 100 / 1000 rule-of-thumb for genetic viability rather than the more normal 50 / 500 rule, and by considering two sub-populations within Finland as being separate units despite the fact that the establishment and separation of these two units has happened on much shorter time scales than those considered for viability assessments.

Estimates of the number of wolves necessary to reach demographic viability (<5% extinction risk over 100 years) ranged from 100 to 300 depending on the extent that immigration from Russia is considered. Estimates of the numbers of wolves needed for short-term genetic viability were 309-376 in western Finland and 187-233 in eastern Finland. The importance of connectivity with Russia and Scandinavia emerged as a central theme from all analysis. The report does not conclude about exact values of FRVs, but provides a basis for political decisions based on decisions about permissible degrees of uncertainty.

Source: Mäntyniemi et al. (2022).

Wolves in Sweden

Sweden (in cooperation with Norway with which it shares the Scandinavian wolf population) has invested heavily in a series of field studies to obtain data on demographics and genetics of the recovering wolf population. There have been multiple processes to analyze this data to inform decisions about FCS and associated FRV values (Bruford 2015, Chapron et al. 2012, Ebenhard 1999, 2000, Nilsson 2004). Based on these analyses and various stakeholder and political processes Swedish interpretations of FRPs have fluctuated over the years, with values of 200 wolves, 20 annual reproductions, 450 wolves, 380 wolves, 170 wolves, and 170-270 wolves – with independent scientists suggesting values ranging from 270 to 1600 (Liberg et al. 2015).

One major process ended in 2015 when different research groups presented and discussed results of their respective analysis of the data. There was a broad consensus that the Habitats Directive goals required the achievement of long-term genetic viability (N_e>500) which was estimated to require around 1700 wolves. However, there was no consensus whether this would need to be achieved within the borders of Sweden or could be achieved as part of a wider meta-population embracing Fennoscandia, western Russia and the Baltics. A majority of researchers agreed that achieving a goal of demographic and short-term genetic viability (N_e>50) would be a more realistic goal within Sweden's borders – and that this could be achieved with a wolf population of c. 300 wolves in Sweden plus c. 40 in Norway and under the assumption of 1 effective immigrant per generation - which formed the basis for a recommendation for the FRP. This figure was based on an estimation of 100 wolves being enough to satisfy demographic viability and an estimate of 170 wolves being enough to satisfy the N_e>50 requirement of short-term genetic viability (assuming N_e/N = 0,3). They then doubled this value to reflect the precautionary principle. These values assume one effective immigrant from outside Scandinavia per generation. A target for the maximum permissible inbreeding coefficient was also proposed at 0.2 (Liberg et al. 2015). There was, however, some internal disagreement among the experts with a minority advocating for larger goals, with more responsibility on Sweden to achieve long-term genetic viability within its borders and to pay more attention to ecological aspects of the recovery concept. Mills and Feltner (2015) supported the 300 wolves as a short-term goal, but suggested that a longer term goal should be to build the population to 600 wolves to achieve greater ecological functionality.

The value of 300 wolves as the FRP for Sweden was reported in their 2013-2018 Article 17 report. However, in their updated management plan from 2016 that was extended to 2021 (Anonymous 2016) the FRP was proposed to be downsized to 270. The FRR is defined as the whole of mainland Sweden, but excluding the alpine zone (because of conflicts with semi-domestic reindeer. In independent analyses using different modelling approaches Miller and Dussex (2024) confirmed that a wolf population in the range of 170 – 270 would be both demographically viable and genetically viable in the short-term, again under the assumption of 1-3 effective immigrants per decade. Different documents on the web currently give both values of 270 and 300 as the current FRP.

Source: Anonymous (2015, 2016), Liberg et al. (2015), Miller & Feltner (2015).

Wolverines, lynx and bears in Sweden

Like for wolves, Sweden in cooperation with Norway has invested heavily in research and monitoring of lynx, wolverines and bears since the late 1980's and early 1990's resulting in a solid platform of knowledge on which to base management decisions. They have also invested heavily in political and stakeholder processes to negotiate population goals.

A parliamentary decision (Reinfeldt & Ek 2013) from 2013 decided on Favourable Reference Population for wolverines, lynx and bears. These values were supported by a set of population viability analyses (Nilsson 2013) that modelled both demographic and genetic aspects of viability, under multiple scenarios involving different degrees of uncertainty and with, and without, catastrophic events. The analysis was based on the premise that the Habitats Directive required long term genetic viability (i.e. N_e>500) but also on the idea that this was not necessarily to be achieved by Sweden alone so "responsibility" was portioned across the Fennoscandian countries to illustrate the potential contribution of each. No politically binding transboundary agreements with concrete goals corresponding to these distributed objectives are currently in place. A follow-up PVA (Mills et al. 2018) has explored the viability of wolverines and lynx in greater detail, but with an emphasis of how different lethal control strategies might influence the viability of these populations.

Wolverines: The PVAs (Nilsson 2013) suggested a need for an overall population of 1378 wolverines for long term viability, which when distributed across the 3 countries and rounded upwards produced a tentative FRP of 500 wolverines for Sweden. This also corresponds to the estimated population when Sweden entered the EU in 1995. The government therefore stated that the FRP should be from 500-600 individuals, which was then operationalized by the Environmental Protection agency as 600 for the FRP with a requirement of at least one reproducing individuals immigrating from Finland or Russia every generation (7 years), with an associated FRR of the alpine zone of Sweden with the surrounding forest areas (Anonymous 2016b). The value of 600 was included in the 2013-2018 Article 17 reporting.

Lynx: The PVAs (Nilsson 2013) suggested a need for an overall population of 1821 lynx for long term viability, which when distributed across the 3 Fennoscandian countries and rounded upwards produced a tentative FRP of 800 lynx for Sweden. The estimated population size when Sweden entered the EU in 1995 was around 700. The government stated that the FRP should be between 700 and 1000 lynx, which was then operationalized by the Environmental Protection Agency (Anonymous 2016c) as an FRP of 870 lynx under the condition of at least one immigrant from Finland or Russia every generation (7 years) if the combined Norwegian and Swedish lynx population was less than 1180 individuals. The FRR was set as all of mainland Sweden. The value of 870 was included in the 2013-2018 Article 17 reporting.

Bear: The PVAs (Nilsson 2013) suggested a need for an overall population of 6838 bears for long term viability, which when distributed across the 3 Fennoscandian countries and rounded upwards produced a tentative FRP of 2800 bears for Sweden. The estimated population size when Sweden entered the EU in 1995 was between 950 and 1200. In contrast to the reasoning for wolverines and lynx above, the government chose a value of 1100-1400, with the lower value based on PVA results for short term genetic viability (<5% heterozygosity loss in 100 years), rather than a long term N_e>500 reasoning, but this time placing all responsibility on Sweden alone. This was then operationalized by the Environmental Protection Agency (Anonymous 2016d) as an FRP of 1400 bears under the condition of at least one immigrant from Finland or Russia every generation (10 years) if the combined Norwegian and Swedish bear population was less than 2350 individuals. The FRR was set as all of the four northern counties and parts of 3 western counties. A value of 1090 was included in the 2013-2018 Article 17 reporting.

Source: Anonymous (2016a,b,c,d), Mills et al. 2018, Nilsson (2013), Reinfeldt & Ek (2013).

Wolves in Lithuania

The objectives of the national Wolf Conservation Plan (produced in 2014, updated in 2019) aim to maintain a wolf population between 31 and 62 packs (from 250-500 wolves). Adaptive management of hunting quotas is used to keep the population within these broad limits. It is apparently implicitly assumed that these national goals align with Favourable Reference Population values. The plan also calls for the maintenance of a wide distribution of these packs throughout the country, which again can be assumed to align with Favourable Refence Range values.

Source: Ministry of the Environment of the Republic of Lithuania (2019).

Eurasian lynx and wolves in Latvia

In the 2013-2018 Article 17 reporting cycle Latvia stated the Favourable Reference Population for wolves was 300 wolves, and the Favourable Reference Range was the whole country. No formal reasoning exists behind this FRP beyond the fact that it broadly corresponds to the population size when Latvia entered the

EU in 2004 and it was assessed as being at Favourable Conservation Status at this time. The Latvian wolf action plan for the period 2018-2028 (Ozolins et al. 2017a) contains no concrete target for the desired size of the wolf population beyond stating that goal is to maintain the conservation status as favourable.

Likewise the FRP for lynx was set at 600 in the 2013-2018 reporting, with the whole country set as FRR. These values correspond to the level when the country entered the EU. The 2018-2028 action plan (Ozolins et al. 2017b) doesn't confirm these exact numbers, but states that the goal is to maintain the favourable conservation status.

Source: Eionet and Ozolins et al. (2017a,b).

Brown bears in Slovenia

The Slovenian bear management plan for the period 2020-2030 defines the Favourable Reference Population value at 800 bears (assessed after reproduction in the spring – i.e. including cubs-of-the-year and before any eventual lethal removal). The objective is further specified in terms of maintain the age and sex structure "as natural as possible". This FRP is justified as being (1) greater than the estimated 540 bears present when Slovenian entered the EU in 2004 such that the precautionary principle is satisfied, (2) the statement that "the size of the brown bear population in Slovenia exceeds the thresholds of the minimum abundance necessary to avoid inbreeding", and (3) a desire to avoid exceeding the social carrying capacity (i.e. the tolerance of rural people). The plan also aims to ensure that habitat remains of suitable quality an that connectivity is maintained within Slovenia, and towards neighbouring countries with which it shares the wider Dinaric-Pindos and Alpine populations, but no quantitative values for range are given.

Source: Ministry of Natural Resources and Spatial Planning (undated)

Lynx and brown bears in France

Current action plans are available for both lynx (2022-2026) and brown bears (2018-2028) in France. The status of both species is recognized as being unfavourable. The action plans clearly state the ambition to improve the conservation status of the species. Large amounts of data are presented, including modelling of potential recovery and intensive monitoring of numbers, distributions and conflucts. Many actions are presented to improve population status and reduce conflicts with rural human communities. However, despite this abundance of technical information and insights no concrete targets are presented and no quantitative values of Favourable Reference Values are given.

Source: Ministry of Ecological Transition and Territorial Cohesion 2018, 2022

Wolves in Denmark

Wolves have only recently colonized Denmark (first wolf in 2012, first reproduction in 2017). Danish authorities have not yet set Favourable Reference Values for wolves. However, an official expert group have analysed the potential for wolves in Denmark which sheds some light on the way that experts in a small and recently colonized country envision the future. Their analysis indicates that Denmark would have the potential to host from 11 to 30 pairs or packs (corresponding to 77 – 210 wolves) if all suitable habitat was occupied. They assume that genetic viability issues can only be achieved via connection to the wider Central European population (Germany, western Poland, Benelux countries) and that Denmark should instead focus on achieving demographic viability, which they estimate can be reached with a population of around 100 wolves (13-17 packs)when transferring data from Scandinavian models and adding a safety margin for precaution.

Source: Sunde et al. (2023).

Discussion

Overall there is an increasingly widespread use of model based approaches to set FRVs. However, there is a wide range of rationales behind the choice of specific model structures. There appears to be a widespread understanding that long-term genetic viability is the overarching goal of the Habitats Directive (corresponding to an N_e >500, or >1000 in some cases), however, in all cases it is made clear that this cannot be reached by single countries and that this must be a collective international objective. National level ambition was variously set with respect to (1) demographic viability or (2) shorter-term genetic viability or (3) taking a share of a collective goal of longer-term genetic viability, depending on national preconditions (size, habitat). A new generation of even more complex PVA models are emerging in some countries / regions that will further refine these approaches. There are however a diversity of technical approaches in use and a diversity of assumptions made about key parameters which means that different methods may produce different estimates, that will have more or less equal validity. Model based approaches underline that resulting estimates depend on many of these assumptions and on subjective choices (such as the range of acceptable probabilities of extinction, acceptable levels of inbreeding or time horizons considered). Ultimately the final choice on many of these parameters may become political decisions with national discretion because there is no legal guidance on such details, and scientific best-practices are not unified.

In contrast, there are still some countries that have FRVs that have no obvious basis beyond approximating what they had when entering the EU and assuming that they were at FCS at this point. There are other countries that present FRVs and where there may be a strong scientific basis, but where it is challenging to find the explicit links within policy documents. And many countries lack quantitative FRVs entirely.

Most countries where we could follow action plans and policy documents discussing conservation targets and FRVs were explicit in that goals for large carnivore conservation must balance conservation concerns (viability) and socio-economic concerns (diverse conflicts), such that upper limits are in placed on carnivore population growth because of conflict potential associated with high densities and certain regions.

Appendix 2 Comments from stakeholders and authorities

The draft report was circulated in advance and discussed among the commission's group of experts on reporting under the Habitats Directive on 21st November 2024 in an online meeting. The draft was then circulated among national authorities and comments were received up until the end of December from a total of 16 entities, including national authorities, individual experts and NGOs. The following is an overview of comments and some short replies. Because many questions were repeated I have organised the issues raised so that each is only addressed once. It is therefore not possible to link a set of comments to the entity that provided them. Many concrete comments on typos, suggestions of minor textual changes for clarity etc have been made directly into the draft report and are not mentioned here.

List of commentators

Individual names have been removed from the list of comments for the sake of anonymity, but institutional characteristics and countries are retained for context.

#1. Individual researcher commenting on behalf of several NGOs. Austria.
#2. Representatives of two Austrian nature management authorities.
#3. Two individual researchers providing scientific support to Austrian authorities
#4. WWF
#5. Individual scientist providing scientific support to Benelux countries
#6. Portuguese nature conservation authorities
#7. Polish biodiversity monitoring authorities
#8. Spanish nature conservation authorities
#9. Individual researcher commenting on request of German authorities
#10. Italian nature conservation authorities
#11. German nature conservation authorities
#12. European hunters NGO
13. An Austrian hunters NGO
#14. Wilderness Society
#15 Slovenian nature conservation authorities
#16 Swedish nature conservation authorities

Comments and responses

1. General comments

The respondents made many highly detailed comments, both around the general approach and specific issues, with many of them providing several pages of comments and / or many suggestions directly on the text. Overall the comments from the different individuals and institutions were very divergent. Many welcomed the initiative to develop more harmonised, robust, and specific guidelines that were linked to issues of genetical viability, transboundary cooperation and forward looking recovery goals (rather than simply minimising the risk of extinction). In contrast, a few questioned the need for any new guidance and / or viewed the targets as being much too ambitious and impossible to achieve. While some felt that the new guidelines left issues open such that Member States would exploit the subjectivity to set minimum goals, others feared that the goals may be too ambitious to be politically or socially acceptable. These different positions led to contrasting comments on the guidelines. The major conceptual developments in these new guidelines – namely the pegging of FRVs to genetical concepts like effective population size and the splitting of the FCS concept to mean different things at different scales were not commented on, hopefully implying broad tacit support.
Because of the often contrasting comments it was not possible to accept all suggestions and requests for changes. However, we have made every effort to include as many comments as possible.
In the following sections we list the main issues identified and give our brief responses to how we have responded to them.

2. Timeline

Issue raised: There was a repeated expression of concern about the time required to adopt the new approach, stating that it would be impossible to incorporate it into the 2019-2024 reporting cycle. It was also proposed to create a specialised working group within the Reporting Working Group.
Response: We agree. Not only would implementing these new guidelines require a large amount of technical and scientific analysis, they may also require multi-national coordination, planning and / or negotiation between competent authorities. We have added a new section (section 9.2) that lists the necessary first steps for implementation of these guidelines for the next reporting cycle. However, it should be possible for an external group of experts or the European Environmental Agency or a working group within Reporting experts to use the data and results from the 2019-2024 reporting process to make a first transboundary assessment of the status of European large carnivores as a pilot study to inform the next round of reporting.

3. Species scope

Issue raised: Why is the Iberian lynx not included? What is the definition of a large carnivore?
Response: There is a certain degree of subjectivity in all such groupings. Iberian lynx were excluded for several reasons, (1) they have a very limited distribution in Europe, (2) the issues with their conservation are very specific and contextual, (3) their spatial ecology is very different, using much smaller home ranges and (4) there is already a huge amount of work being done on their conservation and management, including multiple models looking at their recovery goals, viability and FRVs as well as long established cooperations between responsible authorities. In contrast wolves, bears, Eurasian lynx and wolverines are united by (1) natural low densities and high mobility, (2) high degrees of conflict, (3) very similar management issues, (4) a need to promote better international cooperation. Golden jackals are somewhat different ecologically but are also involved into the same conflict and management challenges and have recently shown a massive expansion across Europe, triggering many discussions that need clarity. The general principles in these guidelines could be applied more widely, although there would be a need to adapt them to species' ecologies and scales of movement.

4. **Social carrying capacity**

Issues raised: Many comments focused on the issue of social, cultural and economic considerations and of the need to take these into account in setting FRVs. It was also the topic with the greatest degree of divergence. On one hand, multiple respondents commented that it was essential to explicitly consider these issues in the guidelines and open for setting FRVs that are lower than what could be achieved from a purely ecological position. In other words claiming that the social carrying capacity was lower than the ecological carrying capacity and that considering this was essential to build sustainable relationships with rural people. These statements quoted Article 2(3) of the Habitats Directive "*Measures taken pursuant to this Directive shall take account of economic, social and cultural requirements and regional and local characteristics*" . In their interpretations it is clear that they view the setting of FRVs as a measure to reach overall conservation goals. On the other hand, multiple respondents quoted the same article from the Habitats Directive along with Article 288 of the Lisbon Treaty "*A directive shall be binding, as to the result to be achieved, upon each Member State to which it is addressed, but shall leave to the national authorities the choice of form and methods*" in support of the opposite conclusion that social, economic and cultural issues <u>cannot</u> overrule the ecological criteria. In this interpretation it appears that the respondents are not viewing FRVs as "measures" and "methods". The latter interpretation is also in line with existing guidelines from the European Commission which underline that only "Technical issues" can overrule ecological criteria in setting FRVs.
Response: We explicitly discuss this issue in section 5.4 of the report and also mention it as being a common issue implemented in national management plans (Appendix 1). In section 5.4 we point out the controversy in interpretation of the legal basis of the argument and identify it as an area that requires more scholarship and / or clarification by the Commission or the CJEU. It is also apparent that Article 191(3) of the TFEU (Treaty on the Functioning of the European Union) may also be relevant. We have updated the section, also in light of the opinion of the advocate general with respect to CJEU case 629/23. However, based on the widespread existing practices and on the well documented conflicts associated with large carnivores these guidelines are built on a premise that long-term tolerance for large carnivores and their long-term conservation is not necessarily enhanced by maximising local densities as opposed to promoting wide distributions and interconnected populations – in other words we also consider that social, economic and cultural need to be considered when setting realistic goals.
Issue raised: The issue of how to estimate social carrying capacity was raised.
Response: It is in principle not something that can calculated directly. Rather it is a product of negotiation between stakeholders and within society, and will almost certainly vary across time and across space.
Issue raised: Calculation of FRVs according to these guidelines requires choosing multiple parameters and setting threshold probabilities, and can be done using multiple alternative analytical approaches. The choice of these involves a certain degree of subjectivity. The subjective values chosen locally / nationally can reflect the extent to which issues like rural development and public safety are given priority over ecological issues associated with population / genetic viability.
Response: This is correct. Issues surrounding effective population size are complex and there are multiple approaches as well as many parameters with uncertain values. This does open for a certain subjectivity in choice of approach. Furthermore, the different levels of scientific knowledge for different species and populations will require different approaches in different situations. There is also a legitimate need to allow some Member State discretion. However, while there are multiple valid ways of making these calculations, there are also many invalid ways that fall outside the current best scientific practice such that the degree of subjectivity has limits.

5. **Complexity**

Issue raised. Multiple respondents commented on the complexity of the new guidelines, mentioning the complex concepts and multiple items on the checklists. They felt that it might be hard for competent authorities and stakeholders to operationalise the,
Response. There is no doubt that these new guidelines are complex, but unfortunately there are no simple answers to complex wildlife management issues. However, previous guidelines have also been complex and

non-specific. These new guidelines are at least much more specific. In section 9 we present multiple checklists and rule-of-thumb heuristics to help operationalise these guidelines. The updated version includes new information to try and make this simply.

6. Parameter estimates

Issue raised: The lack of specific values for key parameters and conversion factors was mentioned as an obstacle to implementation.
Response: We understand this critique. Where numbers exist we have presented them. But the fact remains that values for many key parameters and conversion factors are not readily available. Most of these can be calculated from existing data, but doing so was beyond the scope of this report's resources.

7. Updating populations

Issue raised: The starting point for many discussions are the population units identified in the 2008 report on population level management. In light of the expansion of wolves across Europe there are questions concerning if these remain the unit for assessment of FCS or if we should now consider new larger units.
Response: This is mainly a wolf issue because they have undergone the most dramatic expansions. It is true that most populations have now re-established some degree of connectivity with their neighbours, although this may not be enough to constitute the required level of exchange for effective connections. There have also come new barriers in the form of border security fences that have dramatically reduced the connectivity between the Baltic States and Russia / Belarus / Poland and between Hungary and Serbia / Croatia. We have expanded the text in section 7.1 to state that there is a need to revisit wolf populations and include a preliminary suggestion to consider larger units for genetic viability assessment, reducing the 9 wolf populations to 6 units (1) Nordic, (2) Baltic-Central-European, (3) Italian-Alpine, (4) Carpathian, (5) Southeast Europe, (6) Iberian. This is however highly conditional on new scientific data about geneflow and connectivity. Although the goals of wolf conservation may like to see one continuous distribution this will probably not function as a single genetic unit because of isolation by distance and the high degree of habitat fragmentation that forces sub-structuring of the population. Furthermore, such a unit would be unwieldy as a management unit.
There has been less expansion with other species. One respondent mentioned that Alpine bears should b assessed with those in the Dinaric-Pindos population. This makes sense once connection is established. A similar situation could soon exist for the Scandinavian and Karelian wolverine population.

8. Ecological function

Issue raised. Respondents commented that we had both too much, and too little focus on ecological function, and that it would be hard to measure. It was also mentioned that conflicts between large carnivores and hunter harvest of wild herbivores should get more attention.
Response. The issue cannot be ignored because it is mentioned, albeit obliquely, in the Habitat Directive text. Our proposal is an attempt to explicitly address it in an operational way that can be easily measured through distribution of reproductive units. However, we freely admit that this is at best an indirect measure of the extent of ecological function, but there is no practical alternative for regular monitoring in addition to a lack of conceptual understanding of what the term actually means. We have added a mention of the fact that enhanced ecological function may lead to more conflict in section 5.3.

9. The need for a precautionary approach

Issues raised: Multiple respondents expressed concern that any ambiguity in how the guidelines are applied might be used to opt for minimum interpretations of FRVs. They cited the need to follow the precautionary principle in interpreting the guidelines. Examples of issues include; (1) That when a range of values is produced for a parameter estimate – such as with confidence intervals – it should be best practice to at least take the mean / median value or upper value, and not always the lowest. (2) The one effective migrant per generation requirement for connectivity is really a minimum because it assumes that all populations are connected to each other, and doesn't consider the stepping stone situation where populations are only

connected in sequence i.e. populations A to B, B to C, C to D – but where A has no connection to C or D and B has no direct connection to D and C has no direct connection to A. In such cases, which reflect many European populations of large carnivore, there would need to be more. It is also underlined that effective migration doesn't just mean individuals moving and surviving, it also requires them to breed. (3) There was a suggestion to use a 100:500 rule rather than 50:500 rule as a precautionary step to reduce risks of short term inbreeding. (4) It was mentioned that it was risky to depend on gene flow from Russia because of he fact that they fall outside any conservation agreements. (5) It was pointed out that using simulations of geneflow rather than empirical measures of geneflow may lead to false conclusions. However, another respondent also pointed out that detecting all geneflow can be almost impossible in large populations, indicating that a functional surrogate would be to document continuity of the reproductive part of the population. (6) It was also pointed out that the Iberian and Italian wolves represent distinct subspecies – *Canis lupus italicus* and *C. l. signatus* which deserve specific conservation attention.
Response: We agree that these are all valid concerns and we have integrated many of the suggestions into the text by adding a section 7.10 on precautionary concerns and in 7.5 on monitoring. However, many of these issues are also too detailed to include a report of this type.

10. **Hybrids**

Issue raised: Concern was raised that packs of wolf-dog hybrids should not be included in the Ne estimates for wolves.
Response: We agree and have mentioned this in section 7.5.

11. **National and sub-national obligations**

Issues raised: Multiple respondents expressed concern that the ambiguity resulting from providing a certain degree of discretion to national or sub-national authorities might lead to interpretations for minimal large carnivore populations or that there may even be reductions in some local populations as a result of reaching overall FCS status through connection to transboundary populations. It should also be pointed out that other respondents expressed concerns that reaching these FRVs would require so many large carnivores that it was unlikely to be socially or politically acceptable. Specifically, (1) Multiple respondents referred to the "50% of carrying capacity" heuristic proposed by Epstein et al. as a minimum objective for both member states and sub-national administrative units. (2) Countries that sit at the junction between multiple populations felt that it was important to be explicit that their FRV requirements should be viewed as a cumulative national contribution rather than as population specific. (3) It was requested to make it explicit that no MS should be able to lower its population, even if it was above the FRV at the MS level until the whole population had reached its FRV for the POP level. (4) It was requested to transfer the same scaling logic applied to member states to the sub-national levels too.
Response: We have attempted to be more specific in the guidelines in sections 8 and 9 of the report. However, the reality is that Europe is a diverse place, with diverse ecological situations, an the species concerned here are very different. It is impossible to offer universal concrete numbers / parameters / guidance. In contrast, there is both an ecological and a political need to allow discretion at member state (and possibly also at sub-national levels in federal structures) levels for locally adapted implementation and for national / local level democratic structures. The philosophy of these guidelines is to provide very concrete and ambitious objectives at the larger scales (mainly transboundary) which is where the main European level concerns lie. All administrative / political levels have to ensure that these common minimum goals are reached and maintained. But after that contribution to the collective goal is reached we believe that it is only reasonable to allow a certain discretion for national / sub-national entities to decide the level of ambition and the measures used. This is the same philosophy that was articulated in the 2008 Population Approach guidelines and is simply further articulated in these new guidelines.

12. **Guidance on hunting**

Issue raised: One respondent requested specific guidance on ensuring that hunting / lethal control was conducted in a way that did not jeopardise reaching transboundary objectives, including issues like scientific

quota calculations, annual impact assessments and cross-border alignment of quotas as well as real time information exchange.
Response: We agree with these points and have added them under section 7.8

13. **Cross border monitoring**

Issue raised: One respondent mentioned the need to focus more on harmonising cross border monitoring activities and information exchange – asking for real time data exchange and specific forums for planning cross-border management.
Response: We agree have underlined this in section 7.5.

14. **Status of the guidelines**

Issue raised: Questions were raised as to the status of the guidelines.
Response: At present the guidelines represent the intellectual work of the authors and the others who have contributed. They do not have official legal status. As such the ideas represent a recommendation for best practice. This is clearly stated in the disclaimer on the title page. We have also adjusted the language of the document.

15. **FRVs as targets**

Issue raised: Several respondents pointed out that there was some confusion surround the issue of if FRVs should be viewed as realistic targets that can be achieved or if they should represent ideal and ambitious reference points that may not be reached.
Response: We have explicitly chosen to view FRVs as concrete targets that represent the level of conservation ambition that is a community obligation. Beyond this is a matter of national or sub-national discretion. Our argumentation is made clearly in section 6, but the essence is that recent CJEU rulings differentiate between management options above or below FCS which implies that in practice it is necessary for member states to be able to reach their FRVs.

In addition to these thematic issues there were many specific comments made onto the draft text concerning typos, areas that needed clarity, areas where greater precision was needed, suggestions for new references etc.

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Europäisches Parlament
2019-2024

ANGENOMMENE TEXTE

P9_TA(2022)0423

Schutz der Viehwirtschaft und der Großraubtiere in Europa

Entschließung des Europäischen Parlaments vom 24. November 2022 zum Schutz der Viehwirtschaft und der Großraubtiere in Europa (2022/2952(RSP))

Das Europäische Parlament,

– unter Hinweis auf die Mitteilung der Kommission vom 20. Mai 2020 mit dem Titel „EU-Biodiversitätsstrategie für 2030: Mehr Raum für die Natur in unserem Leben" (COM(2020)0380),

– unter Hinweis auf seine Entschließung vom 9. Juni 2021 zu dem Thema „EU-Biodiversitätsstrategie für 2030: Mehr Raum für die Natur in unserem Leben"[1],

– unter Hinweis auf seine Entschließung vom 15. November 2017 zu einem Aktionsplan für Menschen, Natur und Wirtschaft[2],

– unter Hinweis auf die Richtlinie 92/43/EWG des Rates vom 21. Mai 1992 zur Erhaltung der natürlichen Lebensräume sowie der wildlebenden Tiere und Pflanzen (Habitat-Richtlinie)[3],

– unter Hinweis auf das Übereinkommen über die Erhaltung der europäischen wildlebenden Pflanzen und Tiere und ihrer natürlichen Lebensräume (Übereinkommen von Bern)[4],

– unter Hinweis auf das Programm der Kommission zur Gewährleistung der Effizienz und Leistungsfähigkeit der Rechtsetzung (REFIT),

– unter Hinweis auf die Mitteilung der Kommission vom 12. Oktober 2021 mit dem Titel „Leitfaden zum strengen Schutzsystem für Tierarten von gemeinschaftlichem Interesse im Rahmen der FFH-Richtlinie" (C(2021)7301),

– gestützt auf Artikel 132 Absätze 2 und 4 seiner Geschäftsordnung,

A. in der Erwägung, dass in vielen Teilen Europas bestimmte große Raubtiere, insbesondere Wölfe und Bären, die seit geraumer Zeit in diesen Gebieten nicht vorkamen, ihr Verbreitungsgebiet ausweiten oder sich in diesen Gebieten wieder

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1 ABl. C 67 vom 8.2.2022, S. 25.
2 ABl. C 356 vom 4.10.2018, S. 38.
3 ABl. L 206 vom 22.7.1992, S. 7.
4 ABl. L 38 vom 10.2.1982, S. 3.

ansiedeln, was sie in Konflikt mit menschlichen Aktivitäten bringt, insbesondere mit der extensiven Beweidung durch Schafe und Rinder; in der Erwägung, dass den Weidewirtschaft betreibenden Landwirten erhebliche Kosten entstehen, die durch den Raubfraß bei ihren Herden und die großen Unterschiede zwischen den Mitgliedstaaten und Regionen in Bezug auf Maßnahmen und in einigen Fällen fehlende Maßnahmen zur Unterstützung ihrer Landwirte sowie im Hinblick auf öffentliche Mittel, die sie für Entschädigungs- und Anpassungsmaßnahmen zur Verfügung stellen, verursacht werden;

B. in der Erwägung, dass legislative Maßnahmen wie die Habitat-Richtlinie und internationale Verträge wie das Übereinkommen von Bern zur Erholung der Großraubtierpopulationen, einschließlich des Wolfs, des Braunbären, des Eurasischen Luchses und des Vielfraßes, beigetragen haben; in der Erwägung, dass die Anzahl der Großraubtiere in Kontinentaleuropa 2012 9000 Eurasische Luchse, 17 000 Braunbären, 1250 Vielfraße und 12 000 Wölfe umfasste; in der Erwägung, dass die Anzahl der Wölfe einer Bewertung aus dem Jahr 2018 zufolge 17 000 betrug und damit in den letzten zehn Jahren erheblich gestiegen ist[1], und die Zahlen für andere Arten ähnlich sind; in der Erwägung, dass ferner die Gesamtzahl der Wölfe in der EU-27 auf der Grundlage der besten verfügbaren Daten 2022 in der Größenordnung von 19 000 liegen dürfte und im geografischen Europa wahrscheinlich mehr als 21 500 beträgt[2]; in der Erwägung, dass nach einer Bewertung des Erhaltungszustands des Wolfs (*Canis lupus*) in Europa in den letzten zehn Jahren in Europa ein Anstieg der Wolfspopulation um mehr als 25 % gemeldet wurde[3]; in der Erwägung, dass die Weltnaturschutzunion drei von neun Wolfspopulationen, drei von zehn Braunbärpopulationen und drei von elf Eurasischen Luchspopulationen in Europa als nicht gefährdet eingestuft hat; in der Erwägung, dass beide Vielfraßpopulationen in Europa nach wie vor bedroht sind und der Pardelluchs nach wie vor gefährdet ist;

C. in der Erwägung, dass Wolfspopulationen das Potenzial besitzen, jährlich exponentiell um ungefähr 30 % anzuwachsen;

D. in der Erwägung, dass die negativen Auswirkungen der Angriffe auf Nutztiere durch die wachsende Wolfspopulation zunehmen; in der Erwägung, dass Wölfe, insbesondere in dicht besiedelten Gebieten, zunehmend in die Nähe des Menschen kommen;

E. in der Erwägung, dass allein in Österreich die Zahl der von Wölfen gerissenen Nutztiere 2021 um 230 % auf 680 angestiegen ist; in der Erwägung, dass eine ähnliche Entwicklung der Angriffe von Wölfen auch in anderen Mitgliedstaaten zu beobachten ist, dass sich nämlich 2020 die Zahl der gerissenen Nutztiere in Frankreich auf 11 849, in Deutschland auf 3 959, in Tschechien auf 616, in Belgien auf 139 und in der italienischen Region Südtirol auf 98 belief;

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[1] Rote Liste gefährdeter Arten der Weltnaturschutzunion, „Canis lupus (Wolf)“, abgerufen am 23. November 2022.

[2] Sachverständigengruppe „Initiative für die großen Fleischfresser Europas“ der Species Survival Commission der Weltnaturschutzunion für den Ständigen Ausschusses des Übereinkommens über die Erhaltung der europäischen wildlebenden Pflanzen und Tiere und ihrer natürlichen Lebensräume, „Assessment of the Conservation status of the Wolf (Canis lupus) in Europe“, 2. September 2022.

[3] https://rm.coe.int/inf45e-2022-wolf-assessment-bern-convention-2791-5979-4182-1-2/1680a7fa47 Ebenda, S. 2.

F. in der Erwägung, dass das schnelle Anwachsen der Wolfspopulationen und die schnelle Zunahme ihrer Angriffe auf Nutztiere es den nationalen Verwaltungen erschwert, mit den ihnen gegenwärtig zur Verfügung stehenden Instrumenten wirksam und entschieden zu handeln;

G. in der Erwägung, dass Landwirte angesichts dessen, dass sie selbst von Großraubtieren angegriffen werden, verzweifelt sind und sich missverstanden und machtlos fühlen; in der Erwägung, dass den Angriffen von Großraubtieren bereits Menschen zum Opfer gefallen sind;

H. in der Erwägung, dass die meisten Populationen von Großraubtieren in Europa grenzüberschreitend sind; in der Erwägung, dass einzelne Populationen große geografische Verbreitungen in verschiedenen Ländern innerhalb und außerhalb der EU abdecken können, was zu Situationen führt, in denen für ein und dieselbe Population in einer Region ein günstiger Erhaltungszustand festgestellt wird, während sie in einer Nachbarregion nach wie vor streng geschützt werden muss;

I. in der Erwägung, dass die Überwachungsansätze sehr unterschiedlich sind, was zu einer uneinheitlichen Qualität und Quantität der Daten zu Populationen von Großraubtieren führt;

J. in der Erwägung, dass im Rahmen des LIFE-Programms bereits zahlreiche Projekte zur Entschärfung von Konflikten mit Wildtieren und zur Förderung der langfristigen Koexistenz mit Großraubtieren finanziert wurden; in der Erwägung, dass zwischen 1992 und 2019 durchschnittlich 3,6 Mio. EUR pro Jahr für Projekte ausgegeben wurden, die sich im Rahmen des LIFE-Programms auf Maßnahmen zur Begrenzung der Schäden durch Großraubtiere konzentrieren, und weitere 36 Mio. EUR für laufende Projekte bereitgestellt wurden sowie kontextspezifische Leitlinien zur Wirksamkeit von Minderungsmaßnahmen wie Elektrozäune, aktive Schafhut und der Einsatz von Herdenschutzhunden in vielen verschiedenen Regionen der EU; in der Erwägung, dass es zusätzlicher Projekte in Regionen und zu Großraubtierarten bedarf, die im Rahmen der bisherigen Projekte noch nicht abgedeckt wurden;

K. in der Erwägung, dass Nutztiere, insbesondere, wenn sie auf eingezäunten und offenen Weiden gehalten werden, aufgrund der zunehmenden Präsenz von Großraubtieren (je nach den ergriffenen Maßnahmen und ihrer Wirksamkeit) einem höheren Risiko von Raubfraß ausgesetzt sind; in der Erwägung, dass dies insbesondere in Berggebieten und dünn besiedelten Regionen der Fall ist, in denen die Beweidung erforderlich ist, um diesen prioritären Lebensraum zu erhalten; in der Erwägung, dass in einigen dicht besiedelten Gebieten mit wenigen natürlichen Beutearten für Großraubtiere auch ein größeres Risiko für Nutztiere bestehen könnte;

L. in der Erwägung, dass die Einstellung der Öffentlichkeit zu Großraubtieren von Land zu Land und von Interessengruppe zu Interessengruppe sehr unterschiedlich ist, insbesondere in Regionen, in denen Großraubtiere seit längerer Zeit nicht mehr vorkamen; in der Erwägung, dass die Angst vor Angriffen und das Fehlen einer ausreichenden Unterstützung durch die Behörden im Hinblick auf die Verhinderung von Schäden zur rechtswidrigen Tötung geschützter Arten führen könnten;

M. in der Erwägung, dass die Schaf- und Ziegenhaltung, die am stärksten durch Angriffe durch Großraubtiere gefährdet ist, bereits seit mehreren Jahrzehnten aufgrund allgemeinerer sozioökonomischer Gründe wirtschaftlich unter Druck geraten ist; in der

Erwägung, dass diese anfällige Branche durch extensive Beweidung einen ökologischen Mehrwert erbringen kann, indem sie zur Erhaltung der biologischen Vielfalt in offenen Landschaften in vielen Gebieten mit naturbedingten Benachteiligungen oder geringer Fruchtbarkeit, wie z. B. Almen, und zur Bekämpfung von Phänomenen wie Erosion und Waldbränden beiträgt;

N. in der Erwägung, dass traditionelle Alm- und Weidesysteme aufgrund ökologischer, landwirtschaftlicher und sozioökonomischer Herausforderungen zunehmend aufgegeben werden;

O. in der Erwägung, dass im Rahmen von LIFE-Projekten in einigen Regionen der EU vorbeugende Maßnahmen zur Vermeidung von Konflikten aufgrund der Koexistenz von Menschen und Großraubtieren als erfolgreiche Methoden zur Verringerung der Schäden durch Großraubtiere gemeldet wurden; in der Erwägung, dass die Wirksamkeit dieser Maßnahmen jedoch durch die geografischen Gegebenheiten und die Bedingungen vor Ort beeinträchtigt werden könnte; in der Erwägung, dass diese Maßnahmen zu einem Anstieg des Arbeitsaufwands und zu höheren Kosten für die Landwirte führen können, insbesondere in Regionen, in die Großraubtiere zurückkehren oder in die sie sich ausbreiten; in der Erwägung, dass vorbeugende Maßnahmen zur Vermeidung von Konflikten aufgrund der Koexistenz von Menschen und Großraubtieren kombiniert werden können, um ihre Wirksamkeit zu erhöhen; in der Erwägung, dass die Entschädigungszahlungen, die auf nationaler Ebene geregelt sind, innerhalb der EU unterschiedlich sind und nicht immer eine vollständige Entschädigung für den erlittenen Schaden bieten;

P. in der Erwägung, dass der Verlust und die Verletzung von Nutztieren aufgrund von Angriffen durch Großraubtiere nicht nur wirtschaftliche Schäden für Landwirte und Züchter verursacht, sondern auch erhebliche emotionale Folgen für die jeweiligen Eigentümer hat;

Q. in der Erwägung, dass traditionelle Verfahren der Viehwirtschaft mit einem hohen Schutz der Tiere vor Raubtieren, wie der Einsatz von Schäfern und Herdenschutzhunden und die Unterbringung in Nachtunterkünften, um eine unmittelbare und kontinuierliche Überwachung des Weideviehs zu sicherzustellen, in Europa jahrhundertelang angewandt wurden, aber aufgrund der stark gesunkenen Zahl der Raubtierangriffe schrittweise aufgegeben worden sind; in der Erwägung, dass es sich in einigen Regionen aufgrund der Landnutzungsänderungen mit einem stärker multifunktionalen Ansatz in landwirtschaftlichen Gebieten, der zunehmenden Bedeutung des Tourismus und des derzeitigen sozioökonomischen Drucks, dem die Landwirtschaft in der EU ausgesetzt ist, mit einem deutlichen Rückgang der Zahl der Landwirte und unterdurchschnittlichen Löhnen, als schwierig erweisen könnte, auf diese alten Verfahren in großem Umfang zurückzugreifen; in der Erwägung, dass innovative Lösungen gefunden werden müssen, um die moderne Landwirtschaft an das Vorkommen von Wölfen zu gewöhnen;

R. in der Erwägung, dass es einer konstruktiven Koexistenz von Großraubtieren und Viehzucht bedarf, wo sich der Erhaltungszustand von Großraubtieren weiter positiv entwickeln kann, während den Landwirten Instrumente und ausreichende Finanzmittel zur Verfügung gestellt würden, um Angriffe auf Nutztiere zu bekämpfen und zu verhindern; in der Erwägung, dass alle Bewirtschaftungsentscheidungen auf wissenschaftlichen und soliden Daten beruhen und ökologischen, sozialen und wirtschaftlichen Perspektiven Rechnung tragen sollten; in der Erwägung, dass weitere

Gespräche zwischen Interessenträgern und Landwirten in Gebieten erforderlich sein werden, in denen Großraubtiere mehrere Jahrzehnte lang nicht mehr vorkamen, und dass weitere Anstrengungen im Hinblick auf den Austausch bewährter Verfahren erforderlich sein werden, um die Einführung von vorbeugenden Maßnahmen zu unterstützen und Zugang zu Finanzmitteln zu erhalten; stellt fest, dass das zunehmende Vorkommen von Großraubtieren positive Auswirkungen auf das Funktionieren und die Widerstandsfähigkeit der Ökosysteme, die Erhaltung der biologischen Vielfalt und ökologische Prozesse haben kann, was unter anderem zur Regulierung der Populationen wildlebender Huftiere beiträgt; betont ferner, dass insbesondere in Nationalparks das Vorkommen von Großraubtieren zum Erholungswert von Wäldern und zu einem stetig wachsenden Naturtourismus beiträgt;

S. in der Erwägung, dass die Kommission im Oktober 2021 einen neuen Leitfaden zum strengen Schutzsystem für Tierarten von gemeinschaftlichem Interesse im Rahmen der FFH-Richtlinie (Habitat-Richtlinie) herausgegeben hat, was auch Wölfe einschließt, mit dem die Mitgliedstaaten der EU dabei unterstützt werden sollen, die Umsetzung der Habitat-Richtlinie vor Ort zu verbessern und insbesondere für die vollständige, klare und genaue Umsetzung von Artikel 16 der Habitat-Richtlinie zu sorgen;

1. nimmt die positiven Ergebnisse von Maßnahmen zur Erhaltung der biologischen Vielfalt in Bezug auf die Wiederherstellung von Großraubtierarten in der EU zur Kenntnis sowie ihre Auswirkungen auf das Funktionieren und die Widerstandsfähigkeit der Ökosysteme, die Erhaltung der biologischen Vielfalt und der ökologischen Prozesse und die Viehzucht; betont, wie wichtig es ist, für eine ausgewogene Koexistenz von Mensch, Vieh und Großraubtieren zu sorgen, insbesondere in ländlichen Gebieten, und betont, dass anerkannt werden muss, dass Veränderungen in der Population bestimmter Arten zu einer Reihe ökologischer, landwirtschaftlicher und sozioökonomischer Herausforderungen führen können; erkennt an, dass Artikel 2 Absatz 3 der Habitat-Richtlinie bereits ein ausreichendes Maß an Flexibilität für den Umgang mit diesen Synergien und Kompromissen vorsieht und als zweckmäßig erachtet wird; stellt fest, dass diese Flexibilitätsregelungen weiter geprüft werden sollten;

2. bedauert die Auswirkungen, die Angriffe von Großraubtieren auf das Wohlergehen der Tiere haben, darunter Verletzungen, Aborte, eine verminderte Fruchtbarkeit, der Verlust von Tieren oder ganzen Herden sowie der Tod von Schutzhunden, und fordert die Kommission und die Mitgliedstaaten auf, alles in ihrer Macht Stehende zu tun, um zu verhindern, dass Nutztiere leiden und ihnen geschadet wird;

3. fordert die Kommission auf, die Fortschritte bei der Erreichung eines günstigen Erhaltungszustands für Arten auf der Grundlage wissenschaftlicher Erkenntnisse weiterhin zu bewerten, um das Verbreitungsgebiet und die Größe der Großraubtiere, einschließlich ihrer Auswirkungen auf die Natur und die biologische Vielfalt, ordnungsgemäß zu bewerten und zu überwachen; betont, dass der hohen grenzüberschreitenden Mobilität von Arten Rechnung getragen werden muss, da der Erhaltungszustand der verschiedenen Populationen derselben Art von Region zu Region unterschiedlich sein kann; fordert die Kommission und die Mitgliedstaaten auf, die grenzüberschreitende Zusammenarbeit weiter zu intensivieren, und betont, dass die Überwachung durch eine harmonisierte Methodik koordiniert werden sollte, bei der gegebenenfalls transnationale Populationen und (bio-)geografische Regionen berücksichtigt werden; fordert die Kommission auf, Mittel für Studien zur biologischen Vielfalt bereitzustellen, zum Beispiel im Rahmen von Horizont Europa, anhand derer die Karten über die Verbreitung und Populationsdichte von Großraubtieren aktualisiert

werden sollen; fordert die Kommission auf, dafür zu sorgen, dass die Mitgliedstaaten geeignete Überwachungsmethoden für jede der verschiedenen Großraubtierarten Vielfalt anwenden, die die Zusammenstellung hochwertiger, vergleichbarer und standardisierter Daten für eine wirksame Bewertung der Populationszahlen ermöglichen;

4. begrüßt, dass ein Änderungsvorschlag zur Herabstufung des Wolfes (Canis lupus) von Anhang II in Anhang III des Übereinkommens in die Tagesordnung der 42. Tagung des Ständigen Ausschusses des Berner Übereinkommens aufgenommen wurde; betont, dass der Erhaltungszustand des Wolfs auf gesamteuropäischer Ebene eine Herabstufung des Schutzstatus und folglich die Annahme der vorgeschlagenen Änderung rechtfertigt;

5. erkennt an, dass die Angriffe von Großraubtieren in ganz Europa zunehmen, dass sie bereits menschliche Opfer gefordert haben und sich negativ auf die Viehzüchter ausgewirkt haben; betont, wie wichtig es ist, dass die Mitgliedstaaten auch Informationen über Schäden aufgrund von Großraubtierangriffen sammeln und darüber Bericht erstatten; betont, dass eine gute Überwachung der Entwicklung des Schadensaufkommens für die Tierhalter eine Grundvoraussetzung für eine erfolgreiche Politik ist, wobei die Mitgliedstaaten jedoch unterschiedliche Erhebungs- und Überwachungsmethoden anwenden; unterstreicht die Bedeutung standardisierter Berichtsformate und betont, dass dies gleichermaßen für die Überwachung der Wirksamkeit der Programme zur Minderung von Schäden, einschließlich Entschädigung und Prävention, gelten sollte; fordert, dass die Ergebnisse der Überwachung und die verwendete Methodik zeitnah und transparent veröffentlicht werden; betont, dass die Kommission die Datenerhebung koordinieren und die Analysen durchführen sollte;

6. betont, wie wichtig es ist, die Überwachung der Gesundheit wildlebender Tier- und Pflanzenarten zu verbessern, insbesondere in Bezug auf die Hybridisierung von Wolf und Hund, die frühzeitig proaktiv erkannt werden sollte; fordert eine standardisierte Strategie zur Erfassung von Wolfshybriden und einen transparenten Ansatz, auch durch einen allgemeinen grenzüberschreitenden Austausch von DNA-Proben von Wölfen zwischen Forschungseinrichtungen;

7. fordert die Kommission und die Mitgliedstaaten auf, Regionen, in denen es Konflikte gibt, die sich durch diese Koexistenz ergeben, bei der Klärung der Frage zu unterstützen, wie die in Artikel 16 Absatz 1 der Habitat-Richtlinie vorgesehene Flexibilität angemessen und verantwortungsvoll genutzt werden kann; nimmt Kenntnis von dem aktualisierten Leitfaden der Kommission zum strengen Schutzsystem für Tierarten von gemeinschaftlichem Interesse im Rahmen der FFH-Richtlinie, der am 12. Oktober 2021 veröffentlicht wurde[1]; betont, dass es in der Verantwortung der Kommission liegt, die bestehenden Leitlinien zu präzisieren und ihre Leitlinien gegebenenfalls auch zur Auslegung der Artikel 12 und 16 auf dem neuesten Stand zu halten, und fordert die Mitgliedstaaten auf, die bestehenden Leitlinien besser zu nutzen und wirksam zu handeln, um Schäden, die von Großraubtieren verursacht werden, unter Berücksichtigung grenzüberschreitender Populationen zu verhindern, zu mindern und auszugleichen und einen wirksamen rechtlichen und institutionellen Rahmen zu schaffen, um Landwirte und Züchter dabei zu unterstützen, diese Koexistenz zu

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[1] Mitteilung der Kommission vom 12. Oktober 2021 mit dem Titel „Leitfaden zum strengen Schutzsystem für Tierarten von gemeinschaftlichem Interesse im Rahmen der FFH-Richtlinie“ (C(2021)7301).

ermöglichen;

8. fordert die Kommission auf, die wissenschaftlichen Daten regelmäßig auszuwerten, damit der Schutzstatus von Arten geändert werden kann, sobald der gewünschte Erhaltungszustand gemäß Artikel 19 der Habitat-Richtlinie erreicht ist;

9. fordert die Kommission und die Mitgliedstaaten nachdrücklich auf, Möglichkeiten für verschiedene Interessenträger, einschließlich ländlicher Akteure, zu organisieren, um die Auswirkungen von Großraubtieren zu erörtern; fordert sie nachdrücklich auf, Informationen über praktische Lösungen und Finanzierungsmöglichkeiten für Präventivmaßnahmen gegen Angriffe auf Nutztiere bereitzustellen und eine klare Sensibilisierungskampagne durchzuführen; betont, wie wichtig es ist, Plattformen zu entwickeln für Interessenträger, die sich mit der Koexistenz mit Großraubtieren auf EU-, nationaler und lokaler Ebene befassen, wie die EU-Plattform für die Koexistenz von Menschen und Großraubtieren, und den Dialog, den Erfahrungsaustausch und die Zusammenarbeit bei der Bewältigung von Konflikten zwischen Menschen und geschützten Arten zu fördern; fordert die Kommission auf, die Entwicklung koordinierter Ansätze in allen Mitgliedstaaten zu unterstützen;

10. fordert die Kommission auf, über die Auswirkungen der Präsenz von Großraubtieren in Europa auf die Lebensfähigkeit der Viehzucht, die biologische Vielfalt, die ländliche Bevölkerung und den ländlichen Tourismus, einschließlich des Generationenwechsels in der Landwirtschaft, im Zusammenhang mit den sozioökonomischen Faktoren, die sich auf die Lebensfähigkeit der Viehzucht auswirken, Bericht zu erstatten; fordert die Kommission und die Mitgliedstaaten auf, die Auswirkungen zu bewerten, die Angriffe von Großraubtieren auf das Wohlergehen der Tiere sowie auf das Wohlergehen, das Einkommen und die höheren Arbeits- und Materialkosten der Landwirte haben, wobei auch zu berücksichtigen ist, ob Präventivmaßnahmen umgesetzt wurden und wie wirksam sie waren;

11. fordert die Kommission und die Mitgliedstaaten auf, eine solide und umfassende Bewertung aller relevanten Bedrohungen und Belastungen für jede Großraubtierart und ihre Lebensräume auf europäischer Ebene und in jedem Mitgliedstaat zu entwickeln, sei es durch natürliche Ursachen oder durch von den Menschen verursachte Faktoren; fordert die Mitgliedstaaten und die Kommission auf, auch vorrangige Korridore für Großraubtiere zu kartieren und die wichtigsten ökologischen Korridore, Ermittlung von Ausbreitungshindernissen, Straßenabschnitten mit hoher Sterblichkeit und anderen wichtigen Landschaftsmerkmalen im Zusammenhang mit der verstreuten Verbreitung von Großraubtieren zu ermitteln, um eine Zersplitterung des Lebensraums zu vermeiden;

12. betont, dass Viehzuchtbetriebe in Berggebieten, insbesondere im Alpenraum, besonders anfällig für zunehmende Schäden durch Großraubtiere sind; weist darauf hin, dass die Betriebe in diesen Gebieten oft klein und mit hohen Mehrkosten verbunden sind, dass sie jedoch geschützt und gefördert werden sollten, da sie zum Schutz der Berglandschaft und zum Schutz der biologischen Vielfalt in unwirtlichen Gebieten beitragen; weist darauf hin, dass Ökosysteme wie artenreiches Nardusgrassland auf kieselhaltigen Substraten in Berggebieten sowie alpine und subalpine Kalkrasen gemäß der Habitat-Richtlinie besonders erhaltenswert sind; stellt fest, dass diese Lebensräume in Anwesenheit wilder Raubtiere geschaffen wurden, und weist darauf hin, dass ein wesentlicher Faktor für die Erhaltung dieser Gebiete die extensive Beweidung ist, z. B. durch Rinder und Pferde oder durch von Schäfer überwachte Herden; fordert die

Kommission auf, traditionelle landwirtschaftliche Verfahren wie die Weidewirtschaft, die beaufsichtigte Beweidung, die von der UNESCO anerkannte Wandertierhaltung und die Lebensweise der Weidelandwirte durch konkrete Lösungen zu schützen und zu erhalten; stellt fest, dass einige dieser Verfahren von der vorgeschlagenen Liste potenzieller landwirtschaftlicher Verfahren erfasst werden können, die im Rahmen von Öko-Regelungen finanziert werden;

13. fordert die Kommission und die Mitgliedstaaten auf, anzuerkennen, dass die derzeit verfügbaren Präventivmaßnahmen, einschließlich Zäunen und Schutzhunden, die in einigen Regionen der EU erfolgreich sind, zusätzliche finanzielle und arbeitsbedingte Belastungen für die Landwirte mit sich bringen können, nicht immer durch EU- oder nationale Mittel unterstützt werden und je nach den örtlichen Gegebenheiten ein unterschiedliches Maß an Effizienz und Wirksamkeit aufweisen [1,2]; betont in diesem Zusammenhang, dass die finanzielle Unterstützung für Präventivmaßnahmen mit beratender Unterstützung einhergehen sollte, um deren umfassende und rechtzeitige Umsetzung sicherzustellen; hebt hervor, dass die Beschaffenheit des Geländes, die geografischen Gegebenheiten, die bisherige Koexistenz mit Großraubtieren und andere vorherrschende Faktoren wie der Tourismus, der für die betreffenden Gebiete oft von entscheidender Bedeutung ist, bei der Umsetzung von Präventivmaßnahmen und der Erwägung von Ausnahmeregelungen berücksichtigt werden müssen; fordert die Kommission und die Mitgliedstaaten auf, einzuräumen, dass auf der Grundlage wissenschaftlicher Erkenntnisse proaktiv Strategien zur Schadensbegrenzung im Einklang mit der Habitat-Richtlinie entwickelt und angewandt werden müssen, wenn die Populationen von Großraubtieren zunehmen;

14. fordert die Kommission auf, die Fortschritte bei der Verwirklichung eines bestimmten Erhaltungszustands von Arten auf Ebene biogeografischer Regionen und/oder EU-weiter Populationen regelmäßig zu überprüfen, und beharrt darauf, dass die Kommission entsprechend Artikel 19 der Habitat-Richtlinie unverzüglich ein Überprüfungsverfahren entwickelt, damit der Schutzstatus von Populationen in bestimmten Regionen geändert werden kann, sobald der gewünschte Erhaltungszustand erreicht ist;

15. fordert die Kommission und die Mitgliedstaaten auf, die besten Präventivmaßnahmen zur Verringerung von Angriffen und Schäden durch Großraubtiere wissenschaftlich zu ermitteln und zu unterstützen, wobei die regionalen und lokalen Besonderheiten der Mitgliedstaaten zu berücksichtigen sind, und die Landwirte bei der Beantragung dieser Präventivmaßnahmen zu unterstützen, um erfolgreiche Ansätze zu multiplizieren und zu verbreiten; fordert ferner, dass sie wirksam in Beratungs- und Auskunftsdienste einbezogen werden; fordert eine Aufstockung der LIFE-Mittel für Projekte, die darauf abzielen, die Koexistenz mit Großraubtieren zu erreichen und gleichzeitig die Mittel für den Artenschutz aufrechtzuerhalten; fordert, dass kleinen Projekten Vorrang eingeräumt wird, die auf den Austausch und die Entwicklung bewährter Verfahren für die Koexistenz mit großen Raubtieren abzielen, und fordert die Kommission auf, geeignete Anforderungen für die Messung und Berichterstattung über die Wirksamkeit von Schadensbegrenzungsmaßnahmen festzulegen, die in von der EU finanzierten Projekten, wie etwa im Rahmen des LIFE-Programms, untersucht wurden, wobei

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1 Cortés, Y. u. a., „A decade of use of damage prevention measures in Spain and Portugal“, *Carnivore Damage Prevention News*, 2020.

2 Oliveira, T. u. a., „The contribution of the LIFE program to mitigating damages caused by large carnivores in Europe“, *Global Ecology and Conservation*, Bd. 31, 2021.

objektiven und quantitativen Bewertungsmethoden Vorrang einzuräumen ist;

16. fordert die Mitgliedstaaten auf, umfassende Artenaktionspläne oder Erhaltungs- und/oder Bewirtschaftungspläne zu erstellen und umzusetzen, sofern noch keine vorhanden sind, wobei die menschliche Dichte, Landschaftsstrukturen, Viehzucht, der Erhaltungszustand, andere relevante menschliche Tätigkeiten und wild lebende Huftiere zu berücksichtigen sind;

17. betont, dass die Populationen von Großraubtieren regelmäßig überwacht werden müssen, um Erhaltungsmaßnahmen strategisch zu planen, Präventivmaßnahmen zur Verringerung von Konflikten anzuwenden und die Ergebnisse aller Maßnahmen zu bewerten; weist darauf hin, dass die Überwachung auf einer soliden Methodik beruhen, die Beteiligung verschiedener Interessenträger fördern und erleichtern sollte und dass ihre Ergebnisse regelmäßig der Gesellschaft und den wichtigsten Interessengruppen mitgeteilt werden sollten;

18. fordert die Kommission und die Mitgliedstaaten auf, angemessene und langfristige Finanzierungsmöglichkeiten für geeignete Präventivmaßnahmen und eine angemessene Entschädigung der Landwirte zu ermitteln, und zwar nicht nur für Verluste und Kosten, die ihnen infolge der Angriffe durch Großraubtiere entstehen, sondern auch für die ergriffenen Eindämmungsmaßnahmen, um die Koexistenz von Großraubtieren und nachhaltigen Tierhaltungsmethoden sicherzustellen; betont, dass Entschädigungsregelungen, die so konzipiert sind, dass die Viehzucht und das Vorkommen von Großraubtieren keinen Gewinnausfall für die Landwirte zur Folge haben, direkte und indirekte Kosten im Zusammenhang mit Raubtierangriffen decken und im Sinne größtmöglicher Effizienz mit Präventivmaßnahmen einhergehen sollten; hebt hervor, dass sämtliche Verluste von Nutztieren, die durch Großraubtiere, einschließlich Wolfshybriden, verursacht werden, angemessen und umfassend ausgeglichen werden müssen; fordert die Mitgliedstaaten und die Regionen auf, den Zugang zu finanziellen Entschädigungen zu verbessern; fordert die Kommission auf, anzuerkennen, dass die steigende Zahl der Angriffe von Großraubtieren dazu führt, dass auch die Mittel für den Schutz von Haustieren und die Auszahlung von Entschädigungen zunehmen; bedauert, dass sich die Höhe der Entschädigungen, die Tierhaltern nach einem Angriff gezahlt werden, von Mitgliedstaat zu Mitgliedstaat unterscheiden; fordert die Kommission auf, eine Änderung ihrer Agrarleitlinien in Erwägung zu ziehen, um den Ausgleich von Schäden durch große Raubtiere als staatliche Beihilfe zu erleichtern;

19. beauftragt seine Präsidentin, diese Entschließung der Kommission und dem Rat zu übermitteln.

Follow up to the European Parliament non-legislative resolution on the protection of livestock farming and large carnivores in Europe

1. **Resolution tabled pursuant to Rules 132/(2) and (4) of the European Parliament's Rules of procedure**

2. **Reference numbers:** 2022/2952 (RSP) / B9-0503/2022 / P9_TA(2022)0423

3. **Date of adoption of the resolution:** 24 November 2022

4. **Competent Parliamentary Committee:** N/A

5. **Brief analysis/ assessment of the resolution and requests made in it:**

The ongoing recovery of the wolf and (to a lesser extent) of the bear and the lynx is leading to the resurgence of some social conflicts in the concerned areas, in particular with livestock farmers because of the increased risks of livestock damage in the areas concerned. The issue has previously been addressed by the European Parliament through parliamentary questions, petitions, studies and hearings. A large variety of views have been expressed by members of the Parliament, including those supporting the current rules on large carnivores' strict protection and those asking for a change of those rules because of the damages caused by large carnivores to livestock farming in some areas in the EU.

The current resolution reflects these different views. On the one hand, it acknowledges the positive role played by EU nature legislation in the recovery of large carnivores. It supports the Commission's policy on coexistence, asking the Commission and the Member States to ensure long-term funding for both damage prevention and compensation. It acknowledges that the necessary instruments, derogation tools and flexibilities to address conflicts exist under the current legal and policy framework. On the other hand, the text emphasizes the impacts on livestock farming and rural communities. It welcomes and supports the Swiss proposal to downgrade the protection status of wolves under the Bern Convention, by emphasizing that the conservation status of the wolf at pan-European level justifies a mitigation of its protection status. It asks the Commission to carry out additional studies and analyses, as well as an assessment of the effectiveness of preventive measures tested or implemented under LIFE and other EU funding mechanisms. Finally, the resolution asks the Commission to develop an assessment procedure to enable the protection status of populations in particular regions to be amended as soon as the desired conservation status of species has been reached.

6. **Response to the requests in the resolution and overview of the action taken, or intended to be taken, by the Commission**

The return of the wolf is a challenge in several EU regions. Conflicts arise with livestock breeders in areas where the wolf was absent for decades. In such areas, the traditional knowledge and practices on managing and protecting grazing livestock in the presence of large carnivores have been lost since their extermination.The Commission agrees with the need to ensure a balanced coexistence between humans and large carnivores, by exploiting all the available tools under the current legal and policy framework (**paragraph 1**). A key priority to achieve such a coexistence is to prevent and mitigate as much as possible the associated conflicts, using all the available tools, including, when justified, the targeted removal of some specimens of these species.

The Commission agrees with the need to effectively address the problem of predation of livestock (**paragraphs 2, 5, 12, 15 and 18**) and recalls that this has been an important priority of its policy on large carnivores, since the adoption of the Habitats Directive and the establishment of the LIFE programme. It is important to recall that wolves (and other large carnivores) mainly prey on wild ungulates (red deer, roe deer, wild boar, chamois), but may also prey on livestock, mostly sheep, when wild ungulates are not available and sheep are not sufficiently protected. It is important to note that there are around 60 million sheep and 11,4 million goats in the EU and the impact of wolf predation in the EU, according to a study carried out for the European Parliament in 2018, concerns around 20 000 sheep per year. This corresponds to an average of 0.06% of the sheep in the concerned countries. More recent data collected for the Bern Convention show similar levels of overall predation. In some regions in Europe, however, the scale of predation poses a clear challenge.

Member States are invited to make the best use of the available EU and national funding opportunities. The European Agricultural Fund for Rural Development (EAFRD) allows Member States to programme support for preventive actions and investments aimed at mitigating the risk of damages by large carnivores to livestock farming and at helping to resolve conflicts associated with the conservation of protected species. Several Member States are making use of this possibility and have included targeted interventions in their Common Agricultural Policy (CAP) Strategic Plans. As regards the EAFRD, this may cover preventive investments, for instance, into protective fences and costs for their maintenance, purchase of guard dogs and associated costs, training, technical assistance and monitoring, as well as communication and information of the wider public and cooperation between rural actors aimed at reconciling biodiversity conservation with farming. The latter can facilitate the dialogue among stakeholders about the need for reconciling the protection of biodiversity with human activities. The labour cost of livestock guarding is also eligible for support under agri-environmental-climate commitments. These activities can be funded up to 100% of their costs. The Member States can also support farmers' participation in risk management schemes that can compensate severe production losses. In addition, EU state aid rules allow Member States to compensate up to 100% of direct and indirect costs of damages caused by protected species as well as to finance up to 100% of preventive investments.

As regards the animal welfare of livestock animals (**paragraph 2**), the Commission underlines that livestock grazing outdoor are exposed to a number of health risks that differ from the health risks linked to indoor farming, and, unless appropriate management measures are taken by farmers, their normal mortality rate can be significantly high. For example in Ireland (where large carnivores are not present) the estimates of average sheep flock mortality rates range from 5-6% to 20-25%, depending on the age (ewes or lambs) and the environment (lowland or mountain). Causes of health problems and mortality for grazing livestock include infections, parasites, diseases or malformations, accidents, falls, lightening, and attacks by dogs. However, grazing outdoor has also significant positive impacts on health and animal welfare by offering opportunities for locomotion and express normal specific behaviour. In areas with large carnivores, if livestock is insufficiently supervised and protected, predation is an additional risk factor. Adopting the suitable livestock protection measures (e.g. shepherding) and increasing the supervision of the animals could significantly decrease all the above-mentioned causes of injuries and mortality, keeping welfare benefits of grazing for livestock and reducing, at the same time, the impact of large carnivores. The new CAP Strategic Plans offer possibilities to support the relevant practices (e.g. shepherding through dedicated eco-schemes).

The Commission and the Member States regularly assess progress in achieving the conservation status for all species and habitat types covered by the Habitats Directive (**paragraph 3**). This is carried out every six years, under Article 17 of the directive, based on the data collected, analysed and reported by the Member States. Technical guidelines for this exercise have been elaborated by the Commission in close cooperation with the European Environment Agency and experts from national administrations and stakeholders. They provide a framework for Member States to organise their monitoring activities on the ground and report the collected data to the Commission, following agreed formats and criteria. For each species, data are reported *inter alia* on population size and trends, range surface and trend, available species' habitat and trend and future prospects in terms of pressures and threats (**paragraph 11**). For transboundary populations, Member States have the possibility to carry out joint assessments based on their coordinated monitoring systems and agreed assessment criteria. Member States will provide their next reports in 2025. The Commission will then pool all the data from national reports together, with the help of the European Environment Agency and the European Topic Centre on Biological Diversity, and provide an aggregated assessment of species' status and trends for each EU biogeographic region. The results of this EU assessment will be published in 2026 in the 'State of Nature in the EU' report.

In the course of 2023, the Commission as recently indicated by its President, will also carry out an in-depth analysis of all available scientific and technical data, and all other relevant circumstances at hand, in order to assess whether further measures are needed, including for adapting the protection status of species of Community interest based on technical and scientific progress (**paragraphs 8 and 14**).

The Commission recalls that the proposed amendment for the down listing of the wolf (*Canis lupus*) from Appendix II to Appendix III of the Bern Convention has not been adopted by the Standing Committee during its 42nd meeting (**paragraph 4**). On 25 November 2022 the Council of the European Union adopted Decision No. 14861/2/22 opposing the Swiss proposal for down listing the wolf from Appendix II to Appendix III to the Convention. Consequently, the European Union and its Member States' delegation at the Bern Convention meeting acted accordingly and, as the required two third majority was not reached, that proposal was voted down by the Standing Committee.

Within the current framework of the Habitats Directive, Member States do enjoy considerable possibilities to derogate from the prohibitions of the strict protection regime. Article 16 of the directive allows Member States to enact derogations in order to prevent serious damage (in particular to crops, livestock, forests, fisheries and water and other types of property), in the interests of public health and public safety. It also allows this for other imperative reasons of overriding public interest, including those of social or economic nature, or in order to allow, under specific conditions, the taking or keeping of certain specimens of the species enjoying strict protection, in limited numbers specified by the competent national authorities. Therefore, the existing rules on derogations make it possible to balance different interests against the conservation aims of the directive. Member States have the means to take action to derogate to the directive's provisions in order to address the specific challenges they are currently facing in relation to the wolf population, including local conflicts and circumstances, in line with the principle of subsidiarity. The Commission needs to be kept informed of such measures, which need to be in line with the terms and conditions of the directive, but the decision on those measures solely belongs to national or regional authorities. The Commission will not stand in the way of Member States making use of the various derogation possibilities offered to them, under the terms and conditions of the directive.

The Commission stands ready to further assist Member States facing coexistence conflicts, in particular by raising awareness on the Commission guidance on the strict protection of animal species of Community interest, adopted by the Commission in October 2021, which provides clarifications also on the scope and conditions to make use the derogations under Article 16 of the Habitats Directive (**paragraph 7**). A dedicated annex on the wolf (Annex III) has been added to this guidance, with the specific aim to better explain the interpretation and implementation of the strict protection provisions as well as the possibilities for derogations when dealing with this species. Specific recommendations on how to address coexistence conflicts have also been included. In its regular dialogue with the Member States' authorities and stakeholders, the Commission will continue to assist them in addressing any implementation problems related to large carnivores as well as other aspects of the Habitats Directive. At the same time, the Commission will continue to ensure that its guidance and recommendations are kept up to date, including on the interpretation of Articles 12 and 16 of the Habitats Directive (**paragraph 7**).

The Commission is fully aware that the return of large carnivores can be an additional challenge for livestock farmers in areas where these species have long been absent. It is therefore important to adequately support these farmers to address this challenge, and to implement appropriate practices to reduce the risks of predation (**paragraphs 9, 13 and 18)**. Livestock protection measures have proven effective in preventing or significantly reducing predation risks when properly implemented and tailored to the specific context in which they are applied (see for example: relevant guidance elaborated under the LIFE Eurolargecarnivores project; guidance elaborated by the province of Florence; or the field handbook on prevention solutions elaborated under the regional stakeholders' platform in Grosseto). As regards the costs of predation, it is important to ensure an adequate compensation of damages for the affected farmers. This type of public support for both prevention and compensation aims to help the concerned farmers to face the risk of predation on their livestock, to share the burden and costs of the conservation of large carnivores, making coexistence with protected wildlife easier, in coherence with the integration principles and the objectives of both EU environment and agricultural policy. The joint letter sent by Commissioner Sinkevičius and Commissioner Wojciechowski to all EU Ministers for Agriculture and Environment in November 2021 provided a comprehensive picture of the possibilities and tools to address the conflicts associated to the conservation of large carnivores and invited Member States to make the best use of the available national and EU funding sources to support solutions for coexistence.

The new CAP, with more flexibility, allows Member States to design interventions that are most appropriate in each regional context. Based on the needs identified and the intervention strategy, the Member States have included targeted interventions in their plans. Support for preventive actions may be accompanied by support for training and advisory services.

The EU CAP Network also supports exchanges of best practices among stakeholders in this regard. Within the scope of activities of the EIP-AGRI (agricultural European Innovation Partnership), a Focus Group (a temporary group of selected experts focusing on a specific subject, sharing knowledge and experience) met in May and October 2020 to discuss how to promote innovative and sustainable practices to prevent and control wild animal damage on farms while at the same time protecting wildlife (Wildlife and agricultural production). Some of the outputs were ideas for local innovation projects funded by the EAFRD and carried out by "Operational Groups". The Focus Group discussions also helped identify research needs from practice to be taken up by future research projects funded nationally or at EU level. As an example of Operational Group looking into innovative solutions for extensive livestock farming, a Spanish Operational Group has developed a smart fencing system to protect their cattle from wild animals' attacks.

The Commission also supports many relevant actions through the LIFE programme. Examples include the implementation of protection measures for livestock; the setting up of emergency teams; the establishment of volunteer and ambassador networks to assist livestock farmers; and the promotion of a participatory approach, with the active involvement of all parties concerned.

The Commission is engaged since long in organising opportunities for national authorities and different stakeholders to discuss the impacts of large carnivores as well as solutions and financing possibilities for preventive measures against attacks on livestock, notably through an EU stakeholders' platform on large carnivores and regional platforms. In order to find the most suitable coexistence solutions, measures should be elaborated and implemented with the involvement of the stakeholders concerned and should be tailored to the specific local needs and priorities. The above-described approach is fully in line with the requests expressed by the European Parliament in the resolution on the EU Biodiversity Strategy for 2030.

The Commission stands ready to increase its efforts by expanding the geographical coverage and scope of large carnivores' stakeholder's regional platforms to cover the Member States and regions where on-going conflicts focused on wolves' presence are the highest. Better involving the relevant stakeholders in a participatory approach will provide opportunities to help improve the design and implementation of the most suitable preventive measures and other measures on coexistence.

In relation to impacts on tourism (**paragraph 10**), the Commission notes that in several areas of Europe large carnivores offer opportunities for rapidly increasing nature-based tourism (wolf watching, bear watching, photographing, guided walks to observe signs of presence, etc.). Such wildlife tourism can generate direct income and gains in employment in the concerned areas. The development of wildlife tourism can also lead to increased tolerance toward these species at the local level. Moreover, tourism can educate visitors about large carnivores' ecology and coexistence, raising awareness and promoting conservation efforts on a wider scale, including at international level. Considering the growing number of people interested in wildlife tourism, the Commission has supported the elaboration of dedicated guidelines (under the LIFE Wolfalps EU and LIFE Dinalp Bear projects) on how to develop responsible wolf-related and bear-related tourism, in order to avoid potential negative impacts (e.g. disturbance) on the conservation of these species.

The Commission acknowledges that traditional shepherding practices offer the best opportunities for the coexistence with the protected large carnivore species and for the maintenance of biodiversity-rich semi-natural habitats (**paragraph 12**). The Commission therefore considers that Member States should make the best use of the available tools to restore and support these practices, in accordance with the ecological requirements of the concerned habitats and , where appropriate, of modern tools and solutions (such as electric fences or other devices). The multiple values of mountain livestock farming and its contribution to environmental management, conserving biodiversity and cultural heritage is recognised by EU policies. Various policy and financial instruments are available to Member States. These instruments should be used appropriately and in synergy at national and regional level. This includes full and appropriate use of the opportunities provided by the CAP, the LIFE programme and national funds (State aid) for helping farmers and local communities to prevent or cope with the damages.

As regards the CAP, support is provided to farmers in mountain areas through payments for natural or other area-specific constraints (ANC). ANC payments aim at compensating farmers, in total or partially, for the additional costs and income foregone related to the constraints to which the agricultural production is exposed in the delimited area. Such compensation contributes primarily to ensure a fair income and allow farmers to continue agricultural land management in order to prevent land abandonment. Moreover, apart from support for preventive investments, under agri-environmental-climate management commitments, payments in environmentally beneficial grazing areas may cover for example additional labour costs for the maintenance of protective fences as well as for the maintenance of guard dogs. Under the EU CAP Network, knowledge exchange, innovation and EIP (European Innovation Partnership) implementation is supported with activities such as Focus Groups and Operational Groups or seminars and workshops. In particular, the EIP-AGRI Focus Group: High Nature Value (HNV) - Farming profitability investigated the main socio-economic threats to and the main opportunities for the continued existence of HNV farming.

In relation to the identification of and support for preventive measures to reduce attacks and the damage of predation of livestock by large carnivores (**paragraph 15**), the Commission recalls that under the LIFE programme almost 140 projects funded with EUR 166 million have been implementing activities ranging from livestock protection measures to the setting up of intervention/ emergency teams. LIFE projects also contribute significantly to supporting the active involvement of all the concerned actors through a participatory approach in order to find coexistence solutions. Projects address local problems of coexistence taking into account the regional and local context while considering best practices and lessons learnt from other regions and countries. There has been a steady increase in transboundary projects because most large carnivore populations in Europe are transboundary. Under the LIFE Multi-Annual Work Programme 2021-2024, large carnivores are a policy priority under the sub-programme for Nature and Biodiversity. The amount assigned to the sub-programme Nature and Biodiversity is linked to the annual budget allocation in line with the Multiannual Financial Framework. The LIFE programme uses a bottom-up approach and funds are not pre-allocated to specific topics. Independently from the allocation for nature and biodiversity, the uptake of projects on large carnivores will always depend on the quality of the related project proposals. LIFE projects are selected based on their impacts and replication potential. Applicants must include a strategy for the successful replication and/ or transfer of project solutions and results elsewhere. They need to include tasks that will pave the way towards the multiplication of the impact of the project's solutions and mobilise a wider uptake. This ensures that the successful measures are replicated.

The Commission will also launch a Horizon Europe Call (HORIZON-CL6-2024-FARM2FORK-01-1) on "Agro-pastoral/outdoor livestock systems and wildlife management". Project results are expected to contribute to: innovative and sustainable practices and tools at landscape level to prevent and control negative consequences of interactions between livestock and wild animals to protect wildlife and pastoral/ outdoor production systems; recommendations/ policy advice on optimal management at EU level of wildlife and agro-pastoral systems; decision-making process on wildlife management and land planning participated by relevant stakeholders; and improved coordination across Europe in terms of wildlife management, surveillance and data collection systems (**paragraph 15**).

In the course of 2023, the Commission will work on the definition of criteria for measuring the effectiveness of damage mitigation measures investigated and/ or implemented in projects funded by the EU, including LIFE (**paragraph 15**).

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Evidenzbasiertes Wildtier-management

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9 Wie lassen sich Nutztierübergriffe durch Wölfe nachhaltig minimieren? – Eine Literaturübersicht mit Empfehlungen für Deutschland

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9.1 Einleitung

Das Comeback des Wolfes nach Deutschland begann im Jahr 2000 mit der ersten nachgewiesenen Reproduktion nach über 150 Jahren (Kluth et al. 2002). Seither wächst der Bestand und Wölfe breiten sich in immer mehr Bundesländern aus (Reinhardt et al. 2019; DBBW 2020a). Wölfe sind in Deutschland und in weiten Teilen der Europäischen Union streng geschützt und als prioritäre Art in Anhang IV der Fauna-Flora-Habitat-Richtlinie (FFH-RL) aufgeführt. Mit dem anwachsenden Wolfsbestand nehmen auch die Übergriffe auf Nutztiere in Deutschland von Jahr zu Jahr zu. In einem Punkt sind sich Landwirtschaft, Naturschutz und Politik einig: dass Wolfsübergriffe auf Nutztiere nachhaltig minimiert bzw. soweit es geht verhindert werden sollen. Darüber, wie dieses Ziel am besten erreicht werden kann, gibt es jedoch unterschiedliche Ansichten.

Die meisten Schäden durch Wölfe betreffen erfahrungsgemäß kleine Weidetiere wie Schafe und Ziegen. Die wirtschaftliche Situation der Schäferinnen und Schäfer in Deutschland war bereits vor der Rückkehr des Wolfes extrem angespannt; der Schafbestand und die Zahl der schafhaltenden Betriebe ist, wie in vielen anderen europäischen Ländern, seit Jahrzehnten rückläufig (Linnell und Cretois 2018). Der Großteil der Einnahmen von schaf- und ziegenhaltenden Betrieben kommt aus Zulagen und Zuschüssen für die Landschaftspflege, erst an zweiter Stelle stehen Erlöse aus dem Verkauf von Fleisch; Wolle hat kaum noch eine wirtschaftliche Bedeutung (BLE 2022). Mit der Rückkehr der Wölfe wird die wirtschaftliche Lage von Weidetierhaltenden noch schwieriger. Selbst wenn die Materialkosten von Herdenschutzmaßnahmen in vielen Bundesländern bis zu 100 % gefördert werden (DBBW 2021), so ist der Herdenschutz i. d. R. mit einem erhöhten Arbeitsaufwand verbunden, der noch nicht überall finanziell ausgeglichen wird.

Seit Jahren werden Forderungen immer lauter, dass der Wolfsbestand jagdlich reguliert werden sollte. Als Grund für die Notwendigkeit der Bejagung von Wölfen werden in der Regel Sicherheitsaspekte für den Menschen, die Verringerung von Übergriffen auf Nutztiere und eine verbesserte Akzeptanz von Wölfen angegeben (u. a. AfD 2015; Landkreis Bautzen 2017; CDU/CSU 2018; SMUL 2018; WELT 2018; Deutscher Bundestag 19/584, 19/594). Der entsprechende Druck auf die Politik wird von Jahr zu Jahr stärker. Zumindest jedoch, so die einschlägigen Forderungen, sollten diejenigen Wölfe leichter und schneller zum Abschuss freigegeben werden, die Nutztiere töten. Doch sind solche Maßnahmen wirklich zielführend und helfen sie den betroffenen Weidetierhaltenden unmittelbar und auch längerfristig?

Die Entschärfung von Konflikten zwischen Menschen und Wildtieren sollte evidenzbasiert erfolgen und sowohl menschlichen Werten als auch dem Artenschutz Rechnung tragen (van Eeden et al. 2018a). Der Erhalt der Weidetierhaltung ist, genau wie der Schutz des Wolfes, ein gesamtgesellschaftliches Anliegen. Auch aus Tierschutzgründen ist es geboten, Weidetiere vor Übergriffen durch Beutegreifer zu schützen (§ 3 Abs. 2 TierSchNutzV). Im vorliegenden Beitrag untersuchen wir anhand einer umfassenden Literaturrecherche, ob und unter welchen Bedingungen Wolfsabschüsse wirkungsvoll sind, um Übergriffe auf Nutztiere langfristig zu vermindern. Wir geben einen Überblick über den aktuellen Wissensstand zu folgenden Themen: 1) Warum töten Wölfe überhaupt Nutztiere? 2) Gibt es einen einfachen Zusammenhang zwischen der Anzahl Wölfe und den wolfsverursachten Nutztierschäden, und wie ist die diesbezügliche Datenlage in Deutschland? 3) Wie wirksam sind folgende Managementmaßnahmen hinsichtlich einer nachhaltigen Reduktion von Nutztierschäden: A) eine Bejagung von Wölfen, B) die selektive Entnahme von einzelnen schadensverursachenden Wölfen, C) nicht-letale Herdenschutzmethoden? Abschließend legen wir Empfehlungen zu einem evidenzbasierten und lösungsorientierten Wolfsmanagement in Bezug auf den Wolf-Nutztierkonflikt vor.

9.2 Warum töten Wölfe Nutztiere?

Nicht jeder Wolf tötet Nutztiere, aber jeder Wolf kann es lernen. Wölfe sind große Karnivoren, die sich überwiegend von Huftieren ernähren. Sie töten dabei vor allem die Tiere und Tierarten, die sie am leichtesten überwältigen können. Bei wehrhaften Wildtieren wie Hirschen, Wildschweinen, Elchen oder Bisons sind das vor allem junge, kranke oder alte Individuen. Bei nicht-wehrhaften Arten wie Rehen scheint die diesbezügliche Selektion weniger stark ausgeprägt zu sein (Wagner et al. 2012). Domestizierte Huftiere, insbesondere die kleineren Arten wie Schafe und Ziegen, sind für Wölfe eine besonders einfache Beute, sofern sie nicht geschützt sind. Wölfe lernen u. a. von ihren Eltern, welche Tierarten als Nahrung infrage kommen (Kojola et al. 2004; Fabbri et al. 2018), und können erstaunlich konservativ in ihrer Beutewahl sein. Das bedeutet, nicht jede potenzielle Beutetierart wird von einem Wolf auch sofort als solche erkannt. Die meisten Wölfe in Deutschland töten z. B. keine Rinder, obwohl diese in der Regel nicht gegen Beutegreifer geschützt und somit für Wölfe potenziell verfügbar sind. Trotzdem werden sie von vielen Wölfen nicht als Nahrung wahrgenommen. Das kann sich ändern, wenn ein Wolf z. B. über Nachgeburten oder ein außerhalb der Koppel liegendes Kalb lernt, dass auch Rinder Beutetiere sind. Wölfe sind auch Opportunisten und können so auf Änderungen in der Verfügbarkeit leicht erlegbarer Beutetiere reagieren (Gable et al. 2016, 2017). So gab es von 2012–2017 ein Rudel in der Königsbrücker Heide (Sachsen), dass die dort häufig vorkommenden Biber als Nahrung nutzte (Wauer 2014). Ähnlich werden Wölfe, die gelernt haben, dass Kälber eine leichte Beute sein können, versuchen, auch diese Nahrungsquelle zu nutzen.

In Bezug auf Übergriffe durch Wölfe auf Schafe hat die Erfahrung in Deutschland gezeigt, dass am Anfang einer Schadenskette häufig ungeschützte oder nicht ausreichend geschützte Schafe stehen. Es ist davon auszugehen, dass ein Wolf mit jedem erfolgreichen Übergriff auf ungeschützte/schlecht geschützte Schafe darin bestärkt wird, dies erneut zu versuchen. Solche Schafe sind eine viel einfachere Beute als flinke Rehe oder wehrhafte Wildschweine und Hirsche. Ist der Anreiz erst da, beginnt sich das Schadenskarussell zu drehen. Ein Wolf mit solcher Erfahrung ist eher motiviert, auch bei geschützten Schafen nach einer Schwachstelle im Herdenschutz zu suchen. In der Regel versuchen Wölfe, Hindernisse wie Zäune durch Unterkriechen/Untergraben zu überwinden. Deshalb ist es bei elektrischen Zäunen besonders wichtig, dass die untere stromführende Litze so niedrig ist, dass ein Unterkriechen verhindert wird. Dass Wölfe Zäune springend überwinden, kommt vergleichsweise selten vor (Reinhardt et al. 2012). Sie können das Überspringen jedoch lernen, zum Beispiel an nicht-elektrischen Zäunen oder an nicht korrekt aufgestellten Elektrozäunen. Vielleicht ist an einem Tag ein Zaun nicht unter Strom oder eine Ecke einer Koppel nicht richtig abgespannt und der eigentlich 90 cm hohe Zaun misst dort nur noch 60 cm. Das ist niedrig genug für einen derart motivierten Wolf, einen ersten Sprung zu den Schafen zu wagen. Schließlich kann ein solches Individuum so auch lernen, über elektrifizierte Schafnetze zu springen, die gegenüber anderen Wölfen bei korrekter Installation einen guten Grundschutz bieten. Wölfe, die immer wieder an schlecht geschützten Schafweiden trainieren konnten, sind durch einfache Schutzmaßnahmen deutlich schwieriger abzuhalten als ihre Artgenossen, die keinen solchen Lerneffekt hatten oder sogar an einem korrekt installierten Zaun einen elektrischen Schlag bekommen haben.

Bisherige Erfahrungen aus Deutschland deuten darauf hin, dass Wölfe nicht von ihren Eltern lernen, Zäune zu überwinden, sondern dass es sich hier um einen individuellen Lernprozess handelt. Entsprechende wissenschaftliche Studien zur Untermauerung der Erfahrungswerte liegen bisher noch nicht vor.

Übergriffe auf Nutztiere sind in der Regel weder räumlich noch saisonal gleichmäßig verteilt. Die saisonale Verteilung der Nutztierschäden zeigt vielerorts einen deutlichen Anstieg der Übergriffe auf Schafe im Spätsommer und Herbst (Iliopoulos et al. 2009; FSW 2022). In diesen Monaten haben die schnell wachsenden Wolfswelpen einen besonders hohen Energiebedarf. Gleichzeitig sind die im Frühjahr geborenen Jungtiere der wilden Huftierarten inzwischen herangewachsen und nicht mehr so leicht zu erbeuten wie noch am Anfang des Sommers. Für Wölfe, die Welpen zu versorgen haben, sind daher einfach zu erbeutende Nutztiere besonders attraktiv.

Schaut man sich die räumliche Verteilung der Nutztierschäden auf einer Karte an, findet man Gebiete mit einer erhöhten Anzahl an Übergriffen (sogenannten Prädations-Hotspots) und gleichzeitig Gebiete, in denen es keine oder nur geringe Schäden gibt (z. B. Dondina et al. 2015; Pimenta et al. 2017; Pimenta et al. 2018; FSW 2021; NLWKN 2021). Hotspot-Gebiete sind in der Regel Gebiete mit einem hohen Anteil an nicht oder schlecht geschützten Nutztieren. Die dort lebenden Wölfe haben entsprechend gelernt, diese Nahrungsquelle zu nutzen. Die Wolfsindividuen, für die in Niedersachsen, Schleswig-Holstein und Thüringen Abschussgenehmigungen aufgrund von gehäuften Nutztierschäden erteilt wurden (nicht in allen Fällen wurden die Genehmigungen vollzogen), haben über eine lange Zeit, oft über Jahre Erfahrungen an ungeschützten bzw. nicht korrekt geschützten Nutztieren erwerben können. So überwand der Rüde GW924m in Schleswig-Holstein das erste Mal einen Zaun, der den Vorgaben der dortigen Richtlinie zur Sicherung von Schafen entsprach, nachdem er zuvor 21 Übergriffe auf nicht geschützte Schafe verübt hatte (SH 2021). Auch die nachträgliche Anwendung empfohlener Herdenschutzmaßnahmen (BfN und DBBW 2019) kann zur Verhinderung weiterer Nutztierrisse führen, wie das Beispiel der Ohrdrufer Wölfin (GW267f) in Thüringen zeigt. Dort führte der Einsatz von empfohlenen Schutzzäunen, Beratungen und die Präsenz von Herdenschutzhunden dazu, dass die Schäden durch GW267f in dem Gebiet, das vorher besonders stark betroffen war, bis auf einen Übergriff auf eine ausgebrochene Schafherde auf null reduziert wurden.

9.3 Mehr Wölfe – mehr Nutztierschäden?

Diese Frage ist nicht neu und wurde in der wissenschaftlichen Literatur in den letzten 25 Jahren intensiv diskutiert. Es gibt keinen einfachen Zusammenhang zwischen dem Ausmaß der Nutztierschäden, der Anzahl der Wölfe und der Anzahl der Schafe (Kaczensky 1996; Gervasi et al. 2020). Für einige Großkarnivoren wurde ein Zusammenhang zwischen Nutztierschäden und Prädatorendichte nachgewiesen, für andere nicht (Herfindal et al. 2005; Hobbs et al. 2012; Mabille et al. 2015; Widman und Elofsson 2018; Dalerum et al. 2020). Schäden durch Großkarnivoren sind in hohem Maße kontextabhängig. Häufig bestimmen andere Faktoren als die Größe regionaler Großkarnivoren- und Nutztierbestände das Ausmaß der Schäden (Dalerum et al. 2020). Ein Faktor kann die Nutztierdichte in der Weidehaltung sein (Grilo et al. 2019; Pimenta et al. 2018), ein anderer, ob Großkarnivoren nur sporadisch oder regelmäßig in einem Gebiet vorkommen (Widman und Elofsson 2018; Mayer et al. 2022). Ebenso kann das Nahrungsangebot eine Rolle spielen. Sind wilde Huftiere rar, so ist der Anreiz für Wölfe besonders hoch, Nutztiere als Nahrungsquelle zu nutzen. In erster Linie hängt jedoch das Ausmaß der Schäden vor allem damit zusammen, wie gut oder schlecht Nutztiere vor Übergriffen geschützt sind (Kaczensky 1996; Stahl et al. 2002; Gula 2008; Blanco und Cortés 2009; Imbert et al. 2016; Linnell und Cretois 2018; Pimenta et al. 2018; Kirilyuk und Ke 2020; Mayer et al. 2022). Dies gilt insbesondere für kleinere Nutztierarten wie Schafe und Ziegen. In Gebieten, in denen Wölfe vermehrt größere Nutztiere wie Rinder und Pferde angreifen, trifft dies ebenfalls zu (Álvares et al. 2014; Pimenta et al. 2017; BfN und DBBW 2019). Dort, wo Wölfe immer präsent waren und Herdenschutz traditionell zur guten fachlichen Praxis gehört, ist daher das Schadensniveau oft geringer als in Gebieten, in welche sie erst in den letzten Jahrzehnten zurückgekehrt sind (Gervasi et al. 2021).

Bevor wir uns den Studien zur Wirksamkeit von letalen und nicht-letalen Herdenschutzmaßnahmen im Detail zuwenden, werfen wir einen genaueren Blick auf die Nutztierschäden in Deutschland. Die Daten aus den vergangenen 20 Jahren zeigen, dass in Bundesländern mit vielen Wolfsterritorien tendenziell mehr Übergriffe auf Nutztiere stattfinden (Abb. 9.1). Allerdings zeigen die Daten eine hohe Variabilität. Selbst wenige Wölfe können hohe Schäden verursachen, insbesondere bei kleineren Nutztieren wie Schafen und Ziegen. So waren 2019 die Anzahl der Übergriffe auf kleine Nutztiere in Sachsen und Schleswig-Holstein

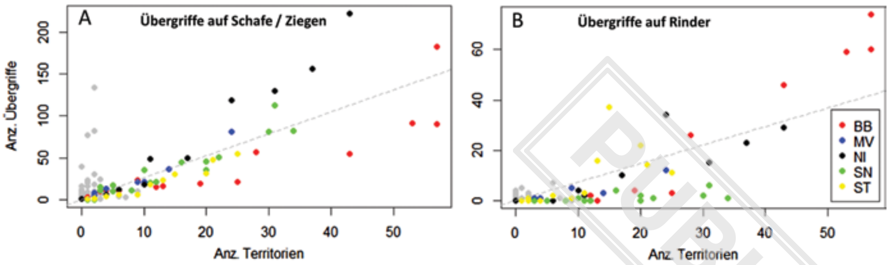

Abb. 9.1 Übergriffe auf Schafe/Ziegen (**A**) und auf Rinder (**B**) für die Jahre 2000–2020 im Verhältnis zur Anzahl Wolfsterritorien in den Bundesländern. Die fünf wolfsreichsten Bundesländer sind farbig dargestellt. Die graue Linie kennzeichnet den durchschnittlichen Zusammenhang zwischen der Anzahl Übergriffe und der Zahl der Wolfsterritorien. Die zugrunde gelegten Daten liefern keine Hinweise, ob und in welchem Umfang die Nutztiere zum Zeitpunkt des Übergriffs durch Herdenschutzmaßnahmen geschützt waren. Nach Angaben der Bundesländer (BB = Brandenburg, MV = Mecklenburg-Vorpommern, NI = Niedersachsen, SN = Sachsen, ST = Sachsen-Anhalt)

Fig. 9.1 Wolf attacks on sheep/goats (**A**) and on cattle (**B**) for the years 2000-2020 in relation to the number of wolf territories in the federal states. The five federal states with most wolf territories are shown in colour. The grey line indicates the average relationship between the number of attacks and the number of wolf territories. The underlying data do not provide any indication whether and to what extent livestock were protected by herd protection measures at the time of the attack. According to data of the federal states (BB = Brandenburg, MV = Mecklenburg-Western Pomerania, NI = Lower Saxony, SN = Saxony, ST = Saxony-Anhalt)

etwa gleich hoch (DBBW 2020b), und das, obwohl Sachsen mit 28 Rudeln und einem Wolfspaar sehr viel mehr Wölfe hatte als Schleswig-Holstein mit 2 territorialen Einzeltieren. Der Schafbestand in Schleswig-Holstein war ca. dreimal so hoch wie in Sachsen (Statistisches Bundesamt 2021). Allein dem ersten territorialen Wolfsrüden in Schleswig-Holstein (GW924m, Pinneberg) konnten in einem Zeitraum von 15 Monaten mehr als 60 Übergriffe auf Schafe zugeordnet werden (SH 2021). Ebenso gibt es große regionale Unterschiede in Bezug auf die betroffenen Nutztierarten. Zum Beispiel sind in Sachsen Übergriffe auf Rinder im Verhältnis zum Wolfsbestand deutlich seltener als in Niedersachsen oder Brandenburg (Abb. 9.1B), obwohl die Größe der Rinderbestände auf der Ebene der Bundesländer vergleichbar ist. Allerdings ist es möglich, dass die räumliche Verteilung der Rinderherden in Weidehaltung innerhalb der Wolfsgebiete unterschiedlich ist.

Schaut man sich die Entwicklungen der Wolfsbestände und der Nutztierschäden für die fünf wolfreichsten Bundesländern genauer an (Abb. 9.2), so fällt auf, dass die Wachstumskurven sehr unterschiedlich verlaufen. Eine nähere Analyse dieser Daten mithilfe einer linearen Regression mit der Anzahl der Übergriffe in Abhängigkeit von der Anzahl der Wolfsterritorien und dem Bundesland (in Interaktion, Abb. 9.3) verdeutlicht zwei Dinge: 1.) Mit der Zunahme der Wolfsterritorien steigen auch die Übergriffe auf Schafe und Ziegen. Für diesen Zusammenhang gibt es eine hohe Evidenz. 2.) Die Stärke des Anstiegs unterscheidet sich zwischen den Bundesländern erheblich. So hat z. B. Niedersachsen im Vergleich zu Brandenburg mehr

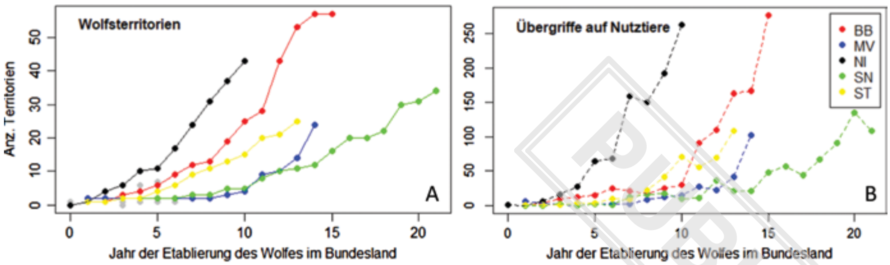

Abb. 9.2 Entwicklung der Anzahl Territorien (**A**) und der Anzahl Übergriffe auf Nutztiere (**B**) im Verhältnis zum Jahr der Etablierung des ersten Wolfsterritoriums in den fünf wolfsreichsten Bundesländern

Fig. 9.2 Evolution of the number of wolf territories (**A**) and the number of attacks on livestock (**B**) in relation to the year of establishment of the first wolf territory in the five federal states with most wolf territories

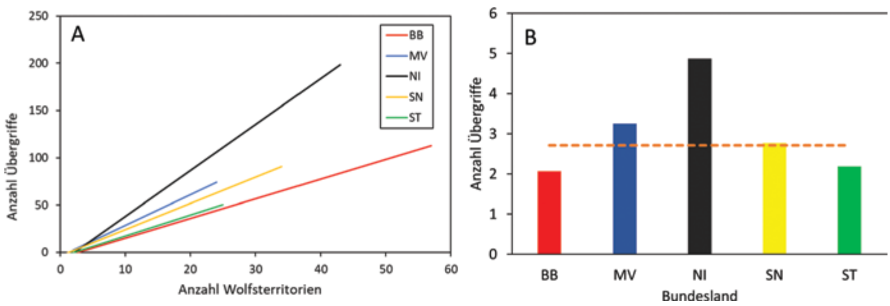

Abb. 9.3 Die auf Basis der realen Wolfs- und Schadenszahlen in einem Modell geschätzte durchschnittliche Entwicklung der Übergriffe auf Schafe/Ziegen in Abhängigkeit von der Anzahl der Wolfsterritorien pro Bundesland (**A**) und die Anzahl der Übergriffe auf Schafe pro Wolfsterritorium in den einzelnen Bundesländern (gestrichelte Linie = modellgeschätzte durchschnittliche Anzahl Übergriffe in den fünf wolfreichsten Bundesländern) (**B**)

Fig. 9.3 The average predicted trend in sheep/goat attacks based on real wolf and damage numbers in a model as a function of the number of wolf territories per state (**A**) and the predicted number of sheep attacks per wolf territory in each state (dashed line = the predicted model-estimated average number of attacks in the five federal states with most wolf territories) (**B**)

als doppelt so viele Übergriffe pro Wolfsterritorium und dementsprechend bei der gleichen Anzahl Wolfsterritorien eine erheblich höhere Schadensbilanz. Die Daten aus Deutschland untermauern somit, dass das Ausmaß der Nutztierschäden nicht allein von der Anzahl der Wölfe abhängt. Auch die Anzahl der vorhandenen Schafe in einem Bundesland hatte keinen nachweisbaren Effekt auf die Anzahl der Übergriffe auf Schafe und Ziegen. Es ist daher zu vermuten, dass die Unterschiede im Schadensniveau zum einen mit der unterschiedlichen räumlichen Verteilung der Schaf- und Ziegenbestände zusammenhängen, vor allem jedoch in der unterschiedlichen Umsetzung von Herdenschutzmaßnahmen in den einzelnen Bundesländern begründet sind.

9.4 Eignung verschiedener Managementmaßnahmen für eine nachhaltige Minimierung von Wolfsübergriffen auf Nutztiere

9.4.1 Bejagung von Wölfen

Als Argument für die Notwendigkeit, Wölfe zu bejagen, wird regelmäßig angeführt, dass dadurch Übergriffe auf Nutztiere reduziert werden können. Schauen wir daher zunächst in Gebiete, in denen der Wolf legal bejagt wird. In Europa sind das innerhalb der EU mehrere Staaten, in denen der Wolf in Anhang V der Fauna-Flora-Habitat-Richtlinie (FFH-RL) gelistet sind, wie z. B. die baltischen Staaten, die Slowakei, Bulgarien und einige Provinzen Spaniens. Allerdings wird der Wolf nicht in allen Staaten/Provinzen, in denen er dem Anhang V der FFH-RL unterliegt, auch bejagt. In Polen ist der Wolf europarechtlich unter Anhang V der FFH-RL gelistet, national ist die Art jedoch streng geschützt und unterliegt nicht dem Jagdrecht. Jährlich werden in Polen aus Managementgründen, z. B. um einer möglichen Gefahr für die menschliche Sicherheit vorzubeugen oder in Gebieten mit einer deutlichen Häufung von Schäden, einzelne Wölfe per Sondergenehmigung zum Abschuss frei gegeben (Reinhardt et al. 2013). In Staaten, in denen der Wolf in Anhang IV der FFH-RL gelistet ist, sind Abschüsse nur im Rahmen von eng definierten Ausnahmeregelungen erlaubt (European Commission 2021).

Versuche, Nutzierübergriffe mittels einer Wolfs-Abschussquote zu reduzieren, gab es in mehreren Ländern. Betrachtet man die wolfsverursachten Nutztierschäden in Europa, so ist nicht erkennbar, dass in Ländern, in denen der Wolf bejagt wird, die Schäden geringer ausfallen als in solchen, in denen dies nicht der Fall ist (vergleiche Daten in Linnell und Cretois 2018). Fernandez-Gil et al. (2016) zeigten für die spanische Provinz Asturias, dass durch legale Abschüsse die durch Wölfe verursachten Nutztierschäden nicht zurückgingen. In einem Vergleich der verschiedenen Wolfs-Managementsysteme spanischer Provinzen und der Höhe der Nutztierschäden gab es ebenfalls keinen Anhaltspunkt dafür, dass in den Provinzen mit Bejagung die Nutztierschäden reduziert würden. Auch die Höhe des Wolfsbestandes war nicht ausschlaggebend für das Schadensausmaß. Viel entscheidender war die Haltungsform der Tiere (Blanco und Cortés 2009). In Slowenien wurde jährlich eine bestimmte Anzahl Wölfe zum Abschuss frei gegeben, mit dem erklärten Ziel, die Nutztierübergriffe zu reduzieren. Allerdings konnte auch nach 15 Jahren der gewünschte Effekt nicht festgestellt werden (Krofel et al. 2011). Das zeigen auch Linnell und Cretois (2018) in einem Vergleich der Nutztierschäden zwischen Norwegen und Schweden: Pro Kopf tötet ein Wolf in Norwegen etwa 40-mal so viele Schafe wie in Schweden – und das, obwohl nur 7 % des norwegischen Schafbestandes innerhalb des dortigen Wolfsgebietes gehalten werden. In Schweden sind es gut 50 % der Schafherden, die sich innerhalb des Wolfsgebietes befinden. Der Hauptunterschied ist, dass schwedische Schafe hinter Zäunen (meist hinter Elektrozäunen) gehalten werden, während norwegische Schafe frei und ungeschützt weiden.

In den USA ist es noch immer gängige Praxis, Wölfe zu töten, um Übergriffe auf Nutztiere zu verhindern (Bergstrom 2017; Vucetich et al. 2017). Mehrere Analysen von wolfsverursachten Nutztierschäden und Wolfsabschüssen in Nordamerika kommen zu dem Schluss, dass Wolfsabschüsse im Zusammenhang mit Nutztierübergriffen, zwar reaktiv, jedoch nicht präventiv sind (Musiani et al. 2005; Harper et al. 2008; Muhly et al. 2010). Das heißt, je höher die Schäden waren, desto mehr Wölfe wurden erlegt. Dies hatte jedoch keinen längerfristigen Effekt und führte nicht zu einer Reduktion der Schäden, weder im selben Jahr noch im Folgejahr.

Kurzfristig kann das Nachstellen durchaus einen Effekt haben. Dieser kann teilweise sogar dann nachgewiesen werden, wenn die Versuche, einen Wolf zu töten, erfolglos waren (Harper et al. 2008; Vogt et al. 2022). Allein die vermehrte Anwesenheit von Menschen an den betroffenen Weiden kann eine abschreckende Wirkung haben und die Übergriffe auf diesen Weiden vorübergehend reduzieren (Stone et al. 2017). Zu ähnlichen Ergebnissen kommt auch eine Untersuchung aus Frankreich (Plisson 2011 zitiert in Grente et al. 2020). In Michigan (USA) führte das Töten von Wölfen auf Farmen mit Nutztierübergriffen zwar in einigen Fällen zu einem Rückgang der Schäden auf diesen Farmen, jedoch stiegen die Schäden auf den benachbarten Farmen gleichzeitig an (Santiago-Avila et al. 2018). Dies spricht für einen Vergrämungseffekt, der das Problem nur räumlich verlagerte, jedoch nicht löste.

Eine weitere nordamerikanische Untersuchung beschäftigte sich mit dem Effekt von Wolfstötungen auf das erneute Wiederauftreten von Nutztierübergriffen über einen Zeitraum von 20 Jahren. Die Autoren kommen zu dem Schluss, dass es [in der Abwesenheit von Herdenschutzmaßnahmen] kaum einen Unterschied gab, ob kein Tier oder mehrere Tiere eines Rudels getötet wurden (Bradley et al. 2015). Laut dieser Studie gab es einen deutlichen Effekt nur dann, wenn das gesamte Rudel eliminiert wurde. Dieser Effekt kam dadurch zustande, dass es mehrere Jahre dauerte, bis sich ein neues Rudel in diesem Gebiet etablierte. Da ein nicht vorhandenes Wolfsrudel auch keine Schäden verursachen kann, ist es genau genommen unzulässig, die Wirksamkeit von Abschüssen als Herdenschutzmethode in einem wolfsfreien Gebiet zu proklamieren (Santiago-Avila et al. 2018), auch wenn der Ausgang für den Tierhalter im Endeffekt der gewünschte war.

Das geplante Entfernen ganzer Rudel ist eine umstrittene Maßnahme, die bei Teilen der Bevölkerung auf große Ablehnung stößt (Eklund et al. 2017) und zudem in Deutschland rechtlich nicht zulässig ist. Zudem ist davon auszugehen, dass es in Gebieten mit flächendeckender Wolfspräsenz nur einige Wochen bis wenige Monate dauert, bis ein vakant gewordenes Territorium von einem neuen Wolfspaar besetzt oder von den Nachbarrudeln übernommen wird (Reinhardt und Kluth 2015).

In einer Meta-Analyse vorhandener Studien stellten Santiago-Avila et al. (2018) fest, dass es bis dato keine wissenschaftlich robuste Studie aus den USA gibt, die belegen konnte, dass das Töten von Wölfen tatsächlich den gewünschten Effekt erzielte, zukünftige Nutztierübergriffe zu verhindern. Solange ungeschützte Nutztiere zur Verfügung stehen, werden neu zugewanderte Wölfe früher oder später ebenfalls dieses Nahrungsangebot nutzen. Seit Jahren fordern daher Wissenschaftlerinnen und Wissenschaftler von den US-Behörden ein evidenzbasiertes Wolfsmanagement, für das Maßnahmen ausgewählt werden, die nach wissenschaftlicher Datenlage tatsächlich dafür geeignet sind, das gewünschte Ziel zu erreichen (Treves et al. 2016; Bergstrom 2017; Vucetich et al. 2017; Santiago-Avila et al. 2018; van Eeden et al. 2018b).

9.4.2 Selektive Einzelabschüsse von Wölfen

Die Antwort des Menschen auf Konflikte mit Wildtieren bestand lange Zeit darin, schadenstiftende Wildtiere wie Bär, Wolf und Luchs stark zu reduzieren oder ganz auszurotten, wie dies in Deutschland geschah. Inzwischen sind die Wiederkehr und der Schutz von Großkarnivoren jedoch gesellschaftlich gewollt, und im Bereich des Managements dieser Tierarten hat ein Umdenken hin zu selektiven Entnahmen stattgefunden (Swan et al. 2017). Mit „Entnahme" ist in der Regel die Tötung eines Tieres gemeint. Solche selektiven Tötungen sollen sich gezielt gegen das schadensverursachende Tier richten. In der wissenschaftlichen Literatur gibt es allerdings nur wenige Beispiele zum Einsatz selektiver Wolfsentnahmen als Mittel zur Reduzierung von Nutztierschäden. Eine Studie aus Montana kommt zu dem Ergebnis, dass im Gegensatz zur allgemeinen Bejagung gezielte Entnahmen durchaus einen Effekt auf das Wiederauftreten von Übergriffen haben (DeCesare et al. 2018), ohne dass jedoch genauer ausgeführt wird, wie diese gezielten Entnahmen durchgeführt wurden. Eine aktuelle Studie konnte zeigen, dass in der Schweiz Einzelabschüsse von schadstiftenden Wölfen die Nutztierschäden kurz- und mittelfristig reduzierten (Vogt et al. 2022). Die Autoren schränken jedoch ein, dass die meisten Fälle aus der frühen Besiedlungsrate der Schweiz stammen, es in den betroffenen Gebieten nach den Abschüssen längere Zeit keine Zuwanderung von Wölfen gab und die Ergebnisse daher nicht ohne Weiteres auf Regionen mit flächiger Wolfspräsenz übertragbar sind.

Vor der Entscheidung, eine Abschussgenehmigung für eine streng geschützte Tierart zu erteilen, muss u. a. geprüft werden, ob es eine zumutbare Alternative dazu gibt (FFH-RL Art. 16/§ 45 BNatSchG). In Deutschland wurde im März 2020 der Paragraph 45 des BNatschG, der die Ausnahmen zur Entnahme streng geschützter Arten regelt, um einen Passus für den Wolf ergänzt. § 45a Abs. 2 BNatSchG enthält nun die rechtliche Grundlage speziell für den Abschuss von Wölfen, die Nutztiere trotz zumutbarer Herdenschutzmaßnahmen wiederholt töten und dadurch ernste landwirtschaftliche Schäden verursachen.

Eine fachliche Definition des für Deutschland **empfohlenen** zumutbaren Herdenschutzes hat das BfN herausgegeben (BfN und DBBW 2019). Zudem erarbeiteten Bund und Länder im Jahr 2021 den „Praxisleitfaden zur Erteilung artenschutzrechtlicher Ausnahmen nach §§ 45 und 45a BNatSchG beim Wolf, insbesondere bei Nutztierrissen", in dem die formell-rechtlichen Anforderungen für die Entnahme von Wölfen betrachtet werden. Dies beinhaltet eine juristische Betrachtung des Kriteriums „Zumutbarkeit". Durch dieses Kriterium soll prinzipiell dem Grundsatz der Verhältnismäßigkeit Rechnung getragen werden. Die letztlich heranzuziehenden

Ökologischer Nachweis	Gezielte Umsetzbarkeit	Bewertung der Wirksamkeit	Verfeinerung	Soziale Bewertung
1. kann das Problem überwiegend einem (oder wenigen) Individuen zugeschrieben werden?	2. Ist es möglich dieses Individuum korrekt zu identifizieren?	3. Wird die gezielte Entfernung dieses Individuums das Problem lösen?	4. Können dadurch unerwünschte Nebeneffekte eintreten? Lassen diese sich minimieren?	5. Kann die Ausrichtung auf das Einzelindividuum helfen, soziale Ziele zu erreichen?
Quantifizierung der individuellen Variation, insbesondere des Einflusses einzelner Individuen.	Einschätzung der Genauigkeit, das schadens-verursachende Individuum von anderen zu unterscheiden.	Anwendung von nicht-letalen Maßnahmen, um deren relative Wirksamkeit gegenüber einer Tötung zu untersuchen.	Einbeziehen von indirekten Effekten (z.B. Einfluss auf restliches Rudel).	Einschätzung, ob durch die Entfernung die angestrebten sozialen Ziele (z.B. Erhöhung der Toleranz) erreicht werden.

Abb. 9.4 Evaluation von selektiven Tötungen mod. nach Swan et al. (2017). Die empfohlenen Phasen der selektiven Tötung sind in der oberen Zeile dargestellt, in der Mitte die Fragen zur Durchführbarkeit, und unten sind die Methoden zur Beantwortung der Fragen aufgeführt
Fig. 9.4 Evaluation of selective killing mod. according to Swan et al. (2017). The recommended stages of selective killing are shown in the top row, the feasibility questions are in the middle, and the methods for answering the questions are listed below

zumutbaren Alternativen müssen jeweils von den Ländern im Rahmen ihrer Zuständigkeit für den Vollzug des BNatSchG festgelegt werden. Allerdings interpretieren die Bundesländer bisher sehr unterschiedlich, was hinsichtlich des Herdenschutzes als zumutbare Alternative anzusehen ist.

Die rechtliche Voraussetzung für die gezielte Entnahme von Wölfen, die nachweislich empfohlene Herdenschutzmaßnahmen überwinden, ist in Deutschland somit vorhanden. Inwiefern eine solche Entnahme praktikabel und zielführend ist, wurde von Swan et al. (2017) näher beleuchtet. Die Autoren entwickelten hierfür einen konzeptionellen Rahmen, der die empfohlenen Abwägungsschritte für die Entscheidung illustriert. Anhand dieses Konzepts kann überprüft werden, ob eine selektive Tötung tatsächlich die geeignete Methode ist, um das Problem zu lösen. (Abb. 9.4).

Im Folgenden wird dieses Schema am Beispiel eines selektiven Abschusses, der zum Ziel hat, die Nutztierschäden in einem Gebiet zu reduzieren, dargestellt. Grundlage für eine entsprechende Abschussgenehmigung wären in diesem Beispiel wiederholte Übergriffe auf Nutztiere trotz empfohlener Schutzmaßnahmen:

Können die Schäden einem oder wenigen Individuen zugeschrieben werden?
Mit den heutigen molekular-genetischen Methoden ist es gut möglich, das schadensverursachende Tier im Labor zu identifizieren. Jährlich werden im Senckenberg Zentrum für Wildtiergenetik tausende nicht-invasive Proben auf Wolfs-DNA untersucht, über die Hälfte davon sind Tupferproben, die an toten Nutztieren genommen werden. An den Bissstellen hinterlässt der Verursacher Speichel und damit Spuren seines Erbguts. Ein Wolf, der wiederholt Nutztiere tötet, wird bei einem entsprechend angelegten Monitoring selbst dann identifiziert, wenn einzelne Tupferproben nicht funktionieren.

Ist es möglich, das Individuum im Feld korrekt zu identifizieren?

Es ist eine Sache, das Individuum im Labor genetisch zu identifizieren und ihm eine eindeutige individuelle Kennnummer zuzuweisen, und eine andere, dasselbe Individuum in freier Wildbahn zu erkennen. Bei anderen Beutegreifern, die einzeln leben, oder solchen mit einem eindeutigen Fellmuster mag dies gelingen. Bei Rudeltieren wie Wölfen ist es schwierig. Selbst wenn bspw. die Elterntiere auf Fotos für Geübte gut von den Jungtieren zu unterscheiden sind, ist die Situation bei Sichtungen im Freiland eine andere. Die Beobachtung ist oft sehr kurz und meist auf größere Entfernung, die Tiere bewegen sich, die Lichtverhältnisse sind i. d. R. suboptimal. Wenn der betreffende Wolf über keine eindeutigen Kriterien der individuellen Erkennbarkeit verfügt (z. B. mit einem Senderhalsband versehen oder mit einem besonderen eindeutigen körperlichen Merkmal ausgestattet), muss die Entnahme in einem engen zeitlichen und räumlichen Zusammenhang mit dem Schadensereignis vollzogen werden, um die Wahrscheinlichkeit zu maximieren, das schadensverursachende Individuum zu töten. Nach erfolgreichen Übergriffen versucht es ein Wolf nicht selten an derselben Herde erneut: In einer Untersuchung aus Schweden war das Risiko eines erneuten Übergriffs in einem Umkreis von etwa 1 km in den ersten drei Wochen nach einem Übergriff besonders hoch (Karlsson und Johansson 2010). Es ist daher zielführend, eine Abschussgenehmigung auf einen engen Radius von etwa 1 km um die betroffene Weide und einen Zeitraum von maximal drei Wochen zu begrenzen, um so die Wahrscheinlichkeit zu maximieren, das schadensverursachende Individuum zu töten.

Wird die gezielte Entfernung dieses Tieres das Problem lösen?

Zunächst muss geprüft werden, ob und inwieweit die Nutztiere vor Wolfsübergriffen geschützt waren. Eine Entnahme wird nur dann den gewünschten nachhaltigen Effekt haben, wenn in dem betroffenen Gebiet ein Wolf trotz fachgerecht geschützter Nutztiere gelernt hat, empfohlene Herdenschutzmethoden zu überwinden, und dieses Individuum dann gezielt getötet wird. Somit kommt der Umsetzung effektiver Herdenschutzmaßnahmen eine maßgebliche Rolle zu. Dagegen wird in einem Gebiet mit vielen ungeschützten Nutztieren ein selektiver Abschuss, wenn überhaupt, nur eine kurzfristige Wirkung haben. Hier ist es nur eine Frage der Zeit, bis der nächste Wolf ungeschützte oder schlecht geschützte Nutztiere tötet. In einer solchen Situation bringt nicht die Wolfstötung die gewünschte nachhaltige Reduktion von Schäden, sondern die flächendeckende Anwendung von wirksamen Herdenschutzmaßnahmen.

Können durch die Entnahme unerwünschte Nebeneffekte eintreten?

Mögliche unerwünschte Nebeneffekte eines Abschusses sollten mitbedacht werden, speziell wenn dadurch eine Veränderung der Rudelstruktur herbeigeführt wird. Handelt es sich um ein Elterntier, das Welpen versorgt, wird sich durch den Abschuss der Druck auf das verbliebene Elterntier erhöhen, ausreichend Nahrung für die Welpen herbeizuschaffen. Das verbliebene Elterntier wird versuchen, an möglichst einfach verfügbare Nahrung zu gelangen. Dadurch erhöht sich in einem Gebiet mit ungeschützten Schafen die Wahrscheinlichkeit, dass durch den Abschuss die Schäden sogar zunehmen (Fernandez-Gil et al. 2016; Fabbri et al. 2018).

Kann die Ausrichtung auf eine Einzelentnahme helfen, soziale Ziele zu erreichen? Verschiedene Bevölkerungsgruppen haben unterschiedliche, teils gegensätzliche Einstellungen gegenüber dem Wolf (Lehnen et al. 2021). Interessengruppen, die direkt und negativ von Wölfen beeinträchtigt werden, fordern häufig eine Bejagung oder vereinfachte Genehmigungsverfahren für Abschüsse (Treves et al. 2013), während naturschutzorientierte Gruppen solchen Maßnahmen kritisch gegenüberstehen (Swan et al. 2017). Im Allgemeinen wird davon ausgegangen, dass die gezielte Entnahme von wenigen Individuen auch bei denjenigen, die eine generelle Bejagung ablehnen, auf größeres Verständnis treffen wird (Linnell 2011; Vogt et al. 2022). Insofern kann die gezielte Tötung eines einzelnen, schadenstiftenden Wolfs ein Kompromiss sein, der beiden Gruppen entgegenkommt.

9.4.3 Nicht-letale Herdenschutzmaßnahmen

Um Übergriffe von Wölfen auf Nutztiere zu reduzieren, sind nicht-letale Herdenschutzmethoden deutlich effektivier als letale Entnahmen (McManus et al. 2014; Miller et al. 2016; Treves et al. 2016; Stone et al. 2017; van Eeden et al. 2018b; Bruns et al. 2020). Farmen in Idaho, USA, die nicht-letale Formen von Herdenschutzmethoden einsetzten, konnten Nutztierübergriffe durch Wölfe dreimal stärker reduzieren als Farmen, welche nur letale Entnahmen anwandten (Stone et al. 2017). Auch die Erfahrungen aus Europa zeigen, dass korrekt angewandte nicht-letale Herdenschutzmaßnahmen den Verlust von Nutztieren durch Großkarnivoren drastisch reduzierten (Linnell und Cretois 2018). Der Effekt ist besonders deutlich, wenn sie gezielt in Prädations-Hotspots eingesetzt werden (Pimenta et al. 2018).

Bewährt haben sich elektrische Zäune, die Kombination von elektrischen Zäunen und Herdenschutzhunden und für frei weidende Schafe eine Behirtung in Kombination mit wolfsabweisenden Nachtpferchen und ggf. Herdenschutzhunden (z. B. Espuno et al. 2004; Salvatori und Mertens 2012; Linnell und Cretois 2018; Ricci et al. 2018; Khorozyan und Waltert 2019; Bruns et al. 2020). Insbesondere bei Rindern ist die Umstellung des Herdenmanagements (Geburtensynchronisation und besonderer Schutz der jungen Kälber) eine weitere erfolgreiche Maßnahme (Breck et al. 2011; Álvares et al. 2014; Pimenta et al. 2017). Als Sofort-Maßnahme sind für einen begrenzten Zeitraum auch rein visuelle Barrieren wie Lappenzäune effektiv (Musiani et al. 2003; Davidson-Nelson und Gehring 2010; Stone et al. 2017; Iliopoulos et al. 2019; Bruns et al. 2020). Inzwischen ist eine Kombination aus Lappen- und Elektrozaun entwickelt worden (sogenannte Turbo-Fladry), durch die Wolfsübergriffe ebenfalls erfolgreich reduziert werden (Lance et al. 2010; Stone et al. 2017). In den zweimal jährlich erscheinenden Carnivore Damage Prevention News (http://www.protectiondestroupeaux.ch/cdpnews/) finden sich viele Fallbeispiele und Erfahrungsberichte aus der Praxis rund um das Thema Herdenschutz.

Allerdings zeigten mehrere Studien, dass insbesondere Elektrozäune häufig nicht korrekt aufgebaut und gewartet werden (Wam et al. 2003 zit. nach Linnell und Cretois 2018; Krofel et al. 2011; Frank und Eklund 2017), wodurch ihre Schutzwirkung eingeschränkt ist. Auch in Deutschland betrifft ein Großteil der wolfsverursachten Schäden nicht oder nicht korrekt geschützte Nutztiere (BfN und DBBW 2019; DBBW 2020b, 2021). So waren in einigen Bundesländern in über 80 % der Wolfsübergriffe auf Schafe keine korrekt angewandten Herdenschutzmaßnahmen vorhanden. Es ist daher unabdingbar, dass Herdenschutzmaßnahmen nicht nur gefördert werden, sondern auch sichergestellt wird, dass sie korrekt angewandt werden (Linnell et al. 2012; Berce und Černe 2020).

Die große Herausforderung besteht darin, Anreize für Tierhaltende zu schaffen, um deren Motivation zu erhöhen, Herdenschutzmaßnahmen korrekt anzuwenden und in Stand zu halten (Linnell und Cretois 2018). Wenn es zu vermehrten Übergriffen auf Nutztiere kommt, brauchen die Betroffenen rasche Abhilfe. Eine sachgerechte Überprüfung der bestehenden Schutzmaßnahmen und die zeitnahe logistische Unterstützung für einen verbesserten Schutz, etwa durch Notfallzaunsets, sollten erste Priorität für amtliche Stellen sein.

9.4.4 Zusammenfassung der Datenlage zur Eignung verschiedener Managementmaßnahmen für eine Minimierung von Wolfsübergriffen auf Nutztiere

Obwohl in vielen Teilen der Welt Großkarnivoren traditionell mit dem Ziel getötet werden, Übergriffe auf Nutztiere zu verhindern (Treves 2019), gibt es keine wissenschaftlichen Belege dafür, dass dadurch die Schäden deutlich verringert werden, es sein denn, der Wolfsbestand wird drastisch reduziert oder ganz ausgelöscht (Bjorge und Gunson 1985; Musiani et al. 2005; Krofel et al. 2011; Linnell und Cretois 2018). Eine generelle Bejagung von Wölfen, ohne sie großflächig auszurotten, ist offensichtlich kein geeignetes Mittel, um Nutztierschäden in Deutschland zu verringern. Getötete Wölfe werden rasch wieder durch Reproduktion oder Neuzuwanderer ersetzt, und auch diese Wölfe werden ungeschützte Weidetiere als Nahrungsquelle entdecken und nutzen, wenn keine geeigneten Herdenschutzmaßnahmen umgesetzt werden.

Auch die häufig geäußerte Idee, bestimmte Gebiete frei von Wölfen zu halten (BSZ 2017; Bauernbund Brandenburg 2018; CDU/CSU 2018), ist nicht zielführend, um Nutztierübergriffe zu reduzieren. Erstens ist dies gegenwärtig mit der rechtlichen Situation unvereinbar (Trouwborst 2018). Zweitens können auch durchwandernde Wölfe erhebliche Schäden an ungeschützten Weidetieren verursachen (Imbert et al. 2016; Mayer et al. 2022). Dies zeigen auch Daten aus Deutschland (vergleiche DBBW 2016–2020; NLWKN 2021; Mayer et al. 2022). Das Schadensniveau kann in einem solchen Gebiet entsprechend hoch bleiben, auch wenn eine Ansiedlung von Wölfen durch kontinuierliche Abschüsse verhindert wird.

Im Gegensatz zu einer undifferenzierten allgemeinen Bejagung des Wolfs kann die gezielte Entnahme von Einzeltieren wirksam sein (Swan et al. 2017). Allerdings haben Einzelfallentnahmen nur dann einen nachhaltigen Effekt, wenn es sich tatsächlich um Individuen handelt, die gelernt haben, funktionstüchtige Schutzmaßnahmen zu überwinden und die sich damit Zugang zu Nutztieren verschaffen, die vor anderen Wölfen in der Region sicher sind. Ansonsten werden auch durch selektive Entnahmen die Schäden, wenn überhaupt, nur vorübergehend verringert (Stahl et al. 2001; Blejwas et al. 2002; Stahl et al. 2002). Zudem ist es sehr schwierig, in der freien Natur ein bestimmtes Individuum sicher zu identifizieren (Linnell et al. 1999; Stahl et al. 2002; Lennox et al. 2018). Dies zeigen auch die bisherigen Erfahrungen aus Deutschland. Selbst in einem Gebiet mit nur einem territorialen Einzelwolf kann sich der Abschuss dieses Tieres als unerwartet schwierig gestalten (Mayer et al. 2022). Um zu gewährleisten, dass mit hoher Wahrscheinlichkeit tatsächlich das schadensverursachende Tier erlegt wird, müssen solche Abschüsse in engem räumlichem und zeitlichem Zusammenhang mit dem Schadensfall stehen (Treves und Naughton-Treves 2005). Sowohl die FFH RL als auch das Bundesnaturschutzgesetz lassen letale Entnahmen nur dann zu, wenn es keine zumutbaren Alternativen gibt. Empfohlene Herdenschutzmaßnahmen werden in aller Regel als zumutbare Alternative betrachtet. In den seltenen Fällen, in denen ein Wolf nachweislich gelernt hat, empfohlene Schutzmaßnahmen zu überwinden, kann eine selektive Entnahme tatsächlich zur Konfliktlösung beitragen.

Die Datenlage der hier ausgewerteten Quellen zeigt eindeutig: Der einzige Weg, um in Koexistenz mit den Wölfen in breiter Fläche eine dauerhafte Reduktion von Schäden an Nutztieren zu erreichen, ist die fachgerechte Umsetzung von Herdenschutzmaßnahmen. Übergriffe auf Nutztiere lassen sich zwar auch dadurch nicht vollständig verhindern. Sie können jedoch durch korrekt angewandte Herdenschutzmaßnahmen deutlich reduziert werden. Die Wirksamkeit von nicht-letalen Herdenschutzmaßnahmen zur Verhinderung von Übergriffen auf Nutztiere ist eindeutig stärker belegt als der Effekt letaler Methoden (Miller et al. 2016; Treves et al. 2016; Eklund et al. 2017; Lennox et al. 2018; Moreira-Arce et al. 2018; van Eeden et al. 2018a, b; Treves 2019). Es gibt daher einen breiten Konsens in der Wissenschaft, dass nicht-letale Methoden zur Verringerung von Nutztierübergriffen durch große Karnivoren nicht nur effektiver, sondern aus ökologischen, rechtlichen und wildtierpolitischen Gründen vertretbarer und gesellschaftlich tolerierter sind als letale Methoden (Bergstrom 2017; Stone et al. 2017; Vucetich et al. 2017; Bruns et al. 2020)

9.5 Der Weg zu einem evidenzbasierten und lösungsorientierten Wolfsmanagement

Wissenschaft und Praxis liefern Daten, Erkenntnisse und Erfahrungen über die gesamte Bandbreite der Managementmaßnahmen, die technisch möglich sind. Jedoch gibt die Wissenschaft nicht vor, was richtig oder falsch, gut oder schlecht ist. So kommt die Wissenschaft nicht **per se** zu dem Schluss, dass es **notwendig** oder sogar angemessen ist, Wölfe zu bejagen (Vucetich et al. 2017). Die Antwort auf die Frage, ob Wölfe bejagt werden sollten, ergibt sich aus den Zielen und Wertevorstellungen der Gesellschaft, die in gesetzlichen Regelwerken ihren Widerhall finden. In diesem Text geht es unter anderem um die Frage, ob der Wolf bejagt werden **muss**, um bestimmte Ziele zu erreichen. Wie gezeigt wurde, ist dies für eine nachhaltige Verringerung der wolfsverursachten Nutztierschäden nicht der Fall.

9.5.1 Klare Zielvorgabe für das Management

Zunächst ist es wichtig, sich zu vergegenwärtigen, was das primäre Ziel einer Managementmaßnahme ist. Dieses sollte klar definiert und dann die dafür passenden Maßnahmen gewählt werden. Auf Basis verfügbarer wissenschaftlicher Erkenntnisse muss bereits vorab evaluiert werden, ob die in Frage kommenden Maßnahmen überhaupt geeignet sind, das Ziel zu erreichen, insbesondere dann, wenn diese auch das Töten von empfindungsfähigen und noch dazu streng geschützten Tieren beinhalten. Um überprüfen zu können, wie wirksam die gewählten Managementmaßnahmen in der konkreten Situation sind, in der sie zum Einsatz kommen, werden Kriterien zur Bewertung des Erfolgs oder Misserfolgs benötigt.

In dem hier dargelegten Beispiel ist das Managementziel eine deutliche und nachhaltige Minimierung der Anzahl von Wolfsübergriffen auf Nutztiere. Das gesamtgesellschaftliche Anliegen, die Weidetierhaltung auch in Wolfsgebieten zu erhalten, lässt sich nur umsetzen, wenn die Tierverluste mit einem vertretbaren Aufwand auf ein Minimum reduziert werden können. Kein Tierbesitzer möchte in der Ungewissheit leben und arbeiten, seine Tiere tot auf der Weide vorfinden zu müssen. Die Tierhaltenden müssen die Gewissheit haben, dass ihre Erwerbsgrundlage durch die Anwesenheit von Wölfen nicht infrage gestellt wird. Sie müssen sich darauf verlassen können, dass die Managementmaßnahmen, die zu einer Minimierung von Wolfsübergriffen auf Nutztiere führen sollen, effektiv sind. Kriterien zur Erfolgskontrolle dieser Managementmaßnahmen können konkrete Werte sein, unter welche die Anzahl der Übergriffe innerhalb einer bestimmten Zeit und Gebietskulisse gesenkt werden sollen (Tab. 9.1). Diese Zielgrößen müssen nicht starr sein. So könnten zum Beispiel in Gebieten, in die Wölfe neu einwandern, für eine Übergangsfrist andere Kriterien gelten als in Gebieten mit jahrelanger Wolfspräsenz. Die gesetzten Zielmarken dienen dem Wolfsmanagement als Erfolgskontrolle. Sie ermöglichen es, gezielt dort nachzusteuern und Ressourcen zu bündeln, wo die anvisierten Zielmarken deutlich verfehlt und damit das Ziel nicht erreicht wurde.

9.5.2 Geld allein hilft nicht

In Deutschland steigen die jährlichen Ausgaben der Bundesländer für Schadensausgleich wie für Präventionsmaßnahmen stetig an (DBBW Schadensberichte 2016–2021). Dies ist in einer schnell wachsenden Wolfspopulation nicht verwunderlich, da neue Gebiete von Wölfen besiedelt und ein immer größer werdender

Tab. 9.1 Prinzip eines lösungsorientierten Wildtiermanagements am Beispiel des Wolf-Nutztier-Konfliktes

Table 9.1 Principle of solution-oriented wildlife management using wolf-livestock conflict as an example

Generelle Vorgehensweise	Beispiel Wolf – Nutztierkonflikt
Identifizieren des Konfliktes	Wölfe töten wiederholt Nutztiere in einem Gebiet.
Formulieren des Ziels	Übergriffe auf Nutztiere in diesem Gebiet deutlich und nachhaltig reduzieren.
Definieren von Messgrößen zur Bewertung des Erfolgs/Misserfolgs der Managementmaßnahme	Übergriffe auf Nutztiere auf ein vorher zu definierendes Maximum, z. B. pro Territorium/Gemeinde/Landkreis/Halter und Jahr senken.
Identifizieren von Maßnahmen, die geeignet sind, um das Ziel zu erreichen	Implementierung von regional angepassten Herdenschutzmaßnahmen (z. B. Elektrozäune, Herdenschutztiere, betriebliche Maßnahmen). Beratung von Nutztierhaltenden.
Erfolgskontrolle/Anpassen der Maßnahmen	Monitoring der Funktionstüchtigkeit von Herdenschutzmaßnahmen. Monitoring der Wirksamkeit von Herdenschutzmaßnahmen. Weiterentwicklung von Herdenschutzmaßnahmen. Bei wiederkehrenden Schäden an Weidetieren, die durch nachweislich empfohlene und umgesetzte Schutzmaßnahmen geschützt waren, selektive Entnahme des Wolfes.

Personenkreis von Nutztierhaltenden damit konfrontiert wird, ihre Tiere vor Übergriffen schützen zu müssen. Die Ausgaben für Herdenschutz lagen zuletzt um das 10- bis 20-fache höher als die Summen für Schadensausgleich (DBBW 2020b, 2021). Dies zeigt, dass die Bundesländer den Schwerpunkt auf die Prävention der Nutztierschäden legen, was ausdrücklich zu begrüßen ist. Mittlerweile fördern alle Bundesländer mit Wolfsterritorien Herdenschutzmaßnahmen.

Allerdings finden Nutztierübergriffe nicht nur in Gebieten statt, in denen Wölfe durchwandern oder die neu besiedelt werden. Auch in Gebieten, in denen bereits seit mehreren Jahren Wölfe leben, kommt es lokal noch immer zu Übergriffen. Teilweise können Prädations-Hotspots über mehrere Jahre bestehen bleiben. Offenbar ist die Auszahlung von Fördergeldern für Herdenschutzmittel allein nicht ausreichend, um die Anzahl der Übergriffe deutlich zu senken. Diese Erkenntnis ist weder neu noch für Deutschland exklusiv. Eine stichprobenartige Überprüfung von geförderten „raubtierabweisenden" Zäunen in Schweden ergab das ernüchternde Ergebnis, dass 86 % Prozent der Zäune nicht funktionstüchtig waren (Frank und Eklund 2017). Ähnliche Erfahrungen gibt es auch aus anderen Ländern (Krofel et al. 2011). Also muss neben den finanziellen Mitteln auch die fachliche Expertise für die korrekte Anwendung und Wartung zur Verfügung stehen und die Funktionstüchtigkeit von Schutzmaßnahmen routinemäßig überprüft werden (Linnell et al. 2012; Khorozyan und Waltert 2019b). Idealerweise entwickelt sich unter den Tierhaltenden eine hohe Fachkompetenz und Eigenmotivation, den Herdenschutz effektiv und sicher umzusetzen. Dies geschieht jedoch nicht von heute auf morgen.

9.5.3 Wirksamkeit von Schutzmaßnahmen prüfen

In der Auswertung der wissenschaftlichen Literatur ist deutlich geworden, dass nicht-letale Herdenschutzmaßnahmen, die darauf abzielen, Wölfen den Zugang zu Nutztieren zu verwehren, wirksamer sind als Abschüsse. Dennoch ist davon auszugehen, dass es zwischen den verschiedenen Schutzmaßnahmen Unterschiede in der Wirksamkeit gibt. Voraussetzung für einen Vergleich der Effektivität verschiedener Herdenschutzmethoden sind Daten zur Funktionstüchtigkeit der im Einsatz befindlichen Maßnahmen. Allerdings gibt es bis heute aus Deutschland, abgesehen von kleinen Einzelfallstudien (Hartleb et al. 2017; Kamp 2021), keine aussagekräftigen Untersuchungen dazu, wie viel Prozent der geförderten Herdenschutzmaßnahmen im Einsatz tatsächlich funktionstüchtig sind. Vor dem Hintergrund stetig steigender Ausgaben für Präventionsmaßnahmen und der prekären finanziellen Situation von Schäferinnen und Schäfern ist diese Diskrepanz schwer verständlich. Wenn nicht klar ist, ob die geförderten Herdenschutzmaßnahmen funktionstüchtig angewandt werden, können auch keine Aussagen zur unterschiedlichen Effektivität verschiedener Herdenschutzmaßnahmen getroffen werden. In der Regel werden Angaben zu vorhandenen Schutzmaßnahmen während der Begutachtung von geschädigten Nutztieren aufgenommen. Allerdings ist es im Nachhinein oft schwierig festzustellen, ob die Schutzmaßnahmen zum Zeitpunkt eines Angriffs ordnungsgemäß funktioniert hat. Daher sind stichprobenartige Kontrollen, unabhängig vom Auftreten von Übergriffen, erforderlich, um ausreichende Daten über die Funktionalität von Präventionsmaßnahmen zu erhalten. Dies ist nicht als Kontrolle der Tierhaltenden zu verstehen, sondern notwendig, um die Ursachen von Übergriffen besser beurteilen und in der Folge abstellen zu können. Solche Kontrollen müssen mit viel Fingerspitzengefühl durchgeführt werden, um Misstrauen oder Ressentiments zu vermeiden. Der Zweck der Schadensverhütung und des Schadensausgleichs besteht schließlich darin, Missstände zu mildern und nicht zu verschlimmern. Andererseits sind Kontrollen gerechtfertigt, wenn Präventionsmaßnahmen mit öffentlichen Geldern finanziert werden. Es ist wichtig, dass Nutztierhaltende verstehen, dass das Ziel darin besteht, die wirksamsten Präventionsmaßnahmen zu ermitteln und so die Schäden zu minimieren. Die informierte Zustimmung zu solchen Kontrollen sollte Bestandteil der Finanzierungsunterlagen und der mit den Nutztierhaltenden unterzeichneten Vereinbarungen sein (Rigg 2022).

Einige Bundesländer führen in ihren Schadensstatistiken an, ob bei einem Wolfsübergriff ein „Mindestschutz“ vorhanden war. Der sogenannte Mindestschutz ist ein Kompromiss zwischen dem Aufwand des Tierhalters und der Sicherheit gegenüber Wolfsangriffen. Die Vorgaben für den Mindestschutz sind entsprechend geringer als für den empfohlenen Herdenschutz und unterscheiden sich teilweise zwischen den Bundesländern. In der Regel ist ein Mindestschutz nur bei Schafen, Ziegen und Gatterwild als Voraussetzung für Ausgleichszahlungen im Schadensfall definiert (Details dazu sind in den jährlichen Berichten zu Prävention und Nutztierschäden ausgeführt: https://dbb-wolf.de/mehr/literatur-download/berichte-zu-praevention-und-nutztierschaeden). Die Angabe, ob ein Mindestschutz vorhanden war, sagt nichts darüber aus, ob ein Zaun zum Zeitpunkt des Übergriffs funktionstüchtig war und ob dieser durch den Wolf oder von den Schafen überwunden wurde. Ein technisch funktionstüchtiger Zaun verliert seine Wirksamkeit, wenn die gekoppelte Fläche zu klein ist und die Schafe darin einer potenziellen Bedrohung nicht ausweichen können und ausbrechen.

Um zu verstehen, warum trotz steigender Präventionsausgaben die Nutztierschäden teilweise auch in Gebieten mit jahrelanger Wolfspräsenz nicht zurückgehen, sind Daten zur Funktionstüchtigkeit der geförderten Schutzmaßnahmen notwendig. Erst auf Basis dieser Daten lassen sich Aussagen zu möglichen Unterschieden in der Wirksamkeit verschiedener Schutzmaßnahmen treffen. Zudem kann durch solche Untersuchungen überprüft werden, ob die Verwendung öffentlicher Mittel für die Förderung von fachlich empfohlenen Herdenschutzmaßnahmen eine sinnvolle Investition darstellt (van Eeden et al. 2018b). Empfehlungen dazu, wie entsprechende Untersuchungen mit einem robusten und zuverlässigen Studiendesign entwickelt werden sollten, finden sich bei van Eeden et al. (2018b), Treves (2019), Treves et al. (2019), Louchouarn et al. (2020), Oliveira et al. (2021). Neben der Untersuchung der rein technischen Aspekte des Herdenschutzes ist es ebenso wichtig herauszufinden, wie die Akzeptanz gegenüber Herdenschutzmaßnahmen bei den Tierhaltenden verbessert und deren Eigenmotivation erhöht werden kann. Auch die effektivste Schutzmaßnahme bleibt nutzlos, wenn sie nicht angewandt wird. Nutztierhaltende sollten daher schon in die Konzeption entsprechender Projekte mit eingebunden werden, um sicherzustellen, dass die Fragen untersucht werden, deren Beantwortung für sie am dringendsten ist. Zudem ist bei den Nutztierhaltenden ein Erfahrungsschatz vorhanden, der unersetzbar ist, um pragmatische, lokal angepasste Herdenschutz-Lösungen zu entwickeln und so Nutztierübergriffe von Wölfen langfristig zu minimieren.

9.6 Fazit

Übergriffe von Wölfen auf Nutztiere werden am effektivsten durch die korrekte Umsetzung von nicht-letalen Herdenschutzmaßnahmen nachhaltig verhindert. Die aktuellen Wolfsverordnungen einiger Bundesländer konzentrieren sich im Kontext von Nutztierschäden jedoch vor allem darauf, wann die in den Ländern geltenden Kriterien für einen Abschuss erfüllt sind (BbgWolfVO 2018; SächsWolfMVO 2019; NWolfVO 2020). Allerdings entsprechen diese Abschusskriterien überwiegend nicht den hier erläuterten Voraussetzungen für selektive Abschüsse. Wie anhand der wissenschaftlichen Evidenz aufgezeigt wurde, ist damit den Tierhaltenden nur selten geholfen, insbesondere wenn nicht die schadensverursachenden Tiere erlegt werden. Solche Maßnahmen, wie auch eine immer wieder geforderte generelle Bejagung, mögen vielleicht ein probates Mittel sein, um politischem Druck zu begegnen, sie sind jedoch nach Auswertung der wissenschaftlichen Literatur keine wirksamen Maßnahmen, um Übergriffe auf Nutztiere zu minimieren und den betroffenen Weidetierhaltenden wirklich zu helfen.

Insbesondere die Berufsschäferinnen und Berufsschäfer in Deutschland sind, unabhängig vom Wolf, schon lange in einer extrem schwierigen wirtschaftlichen Lage. Die Rückkehr des Wolfes ist in dieser Situation für einige der Tropfen, der das Fass zum Überlaufen bringt. Durch den nun permanent erforderlichen Herdenschutz steigt die ohnehin schon hohe Arbeitsbelastung weiter. Wollen wir als Gesellschaft die Weideschafhaltung auch zukünftig erhalten, muss die wirtschaftliche Lage der Tierhaltenden deutlich verbessert werden, unabhängig davon, ob es Wölfe gibt. Die von einigen Bundesländern inzwischen eingeführte Weidetierprämie ist hier ein Schritt in die richtige Richtung, um den durch den Herdenschutz erforderlichen zusätzlichen Arbeitsaufwand finanziell abzupuffern.

Das Wissen, wie Weidetiere vor Übergriffen durch Wölfe geschützt werden können, und das technische Know-how dafür sind heute auch in Deutschland vorhanden. Die Methoden sind seit Jahrzehnten bekannt, auch wenn sie stets weiterentwickelt und den lokalen Bedingungen angepasst werden sollten. Viele Tierhaltende haben hier inzwischen ein hohes Maß an Fachkompetenz entwickelt. In den nächsten Jahren geht es darum, Wege zu finden, wie die Akzeptanz von Schutzmaßnahmen und die Eigenmotivation der Tierhaltenden, diese anzuwenden, erhöht werden kann. Nur durch einen möglichst flächendeckenden fachgerechten Einsatz von Herdenschutzmaßnahmen kann verhindert werden, dass Wölfe an nicht korrekt geschützten Tieren das Überwinden dieser Maßnahmen erlernen. Die Erfahrung aus den vergangenen 20 Jahren zeigen, dass die reine Finanzierung von Präventionsmaßnahmen nicht ausreicht. Um die Funktionstüchtigkeit der eingesetzten Schutzmaßnahmen zu gewährleisten, ist eine routinemäßige Überprüfung notwendig. Daneben sollte jeder Tierhaltende bei Bedarf auch fachliche Expertise zur korrekten Anwendung der Maßnahmen in Anspruch nehmen können. Vor allem in Gebieten mit Prädations-Hotspots sollte aktiv auf die Tierhaltenden zugegangen werden, damit Herdenschutzmaßnahmen konsequent umgesetzt werden.

Um gezielter als bisher auf die steigende Zahl von Übergriffen auf Nutztiere in Deutschland reagieren zu können, werden dringend Daten zur Funktionstüchtigkeit, Anwendbarkeit und Akzeptanz der eingesetzten Herdenschutzmaßnahmen benötigt. Diese Daten sind zudem die Grundlage für wissenschaftliche Studien zu möglichen Unterschieden in der Wirksamkeit verschiedener Herdenschutzmethoden. Entsprechende Untersuchungen sind nur in enger Zusammenarbeit zwischen Weidetierhaltung und Wissenschaft möglich. Der Weg von einem emotionsbasierten hin zu einem evidenzbasierten Wolfsmanagement führt über wissenschaftlich robuste Daten und Analysen, basierend auf der Fachkompetenz und den praktischen Erfahrungen der Weidetierhaltenden.

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The spatial distribution and temporal trends of livestock damages caused by wolves in Europe

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ABSTRACT

Wolf populations are recovering and expanding across Europe, causing conflicts with livestock owners. Here we compiled incident-based livestock damage data across 21 countries for the years 2018, 2019 and 2020, during which 39,262 wolf-caused incidents were reported from 470 administrative regions. We found substantial regional variation in all aspects of the data, including the primary target species, the density of damages, their seasonal distribution, and their temporal trend. More than half of the variation in damage densities across regions was explained by the area of extensively cultivated habitats occupied by wolves, namely natural grasslands and broad-leaved forests. Regional variation in husbandry practices and damage prevention, while difficult to quantify at a continental scale, appear important factors to further modulate these incidents. As illustrated with detailed data from Germany, a relationship between the number of wolf units and damages diminished over time, suggesting some adaptation of livestock owners and local authorities to their presence, for example by increasing prevention efforts. As we argue, temporal trends of damage incidents, which are robust to variation in data collection across regions, are thus informative about the local intensity of the wolf-human conflict. We estimated

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increasing trends for the majority of regions, reflecting the current expansion of wolves across the continent. Nonetheless, many of these increases were moderate and for more than one third of all regions, trends were negative despite growing wolf populations, thus indicating that wolf-livestock conflicts can be successfully mitigated with proper management.

1. Introduction

The last decades have seen the recovery of wolves (*Canis lupus*) across Europe, including several regions where the species had previously been extinct for decades or even centuries (Chapron et al., 2014; Reinhardt et al., 2019). Between 2012 and 2016, an estimated 17,000 wolves roamed the European continent (excluding Russia and Belarus, Boitani et al. 2018) and, with the exception of one isolated population in Spain (López-Bao et al., 2018), all populations are continuing to expand (Chapron et al., 2014; Linnell and Cretois, 2018). This recolonization process is taking place without reintroductions and is due to three main factors. First, wolves are granted strict legal protection in many countries by the EU Habitats Directive and/or the Bern Convention (Chapron et al., 2014; Epstein et al., 2016). Second, populations of important prey species such as roe deer (*Capreolus capreolus*), red deer (*Cervus elaphus*) and wild boar (*Sus scrofa*) were able to recover following land abandonment and reforestation in Europe (Trouwborst, 2010). Third, wolves have a remarkably high adaptive capacity, allowing them to establish in fragmented, human-dominated landscapes (Mech and Boitani, 2007; Trouwborst, 2010; Sazatornil et al., 2016; Cimatti et al., 2021).

Wolf recovery is not exempt from social tensions and conflicts (Dressel et al., 2015; Skogen et al., 2017). While ecologists are regarding the growing wolf population in Europe as a conservation success story, many farmers in recovering areas fear increased depredation of their livestock and, as a consequence, a threat to their livelihoods (van Eeden et al., 2018; Bautista et al., 2019; Rode et al., 2021). It is imperative to address these conflicts and to facilitate the coexistence of humans and wolves to ensure positive conservation outcomes in Europe densely populated by humans. The absence of wolves for an extended period has often resulted in reduced adaptations for coexistence (López-Bao et al., 2017), which in turn harbors potential for conflict once the species is recolonizing its former habitat (Chapron et al., 2014; Gervasi et al., 2021a).

To mitigate these conflicts, authorities aim at raising the standards of livestock protection to reduce livestock vulnerability and shift depredation from livestock to wild prey (van Eeden et al., 2018; Eklund et al., 2017). In Europe, most countries provide financial support to their farmers to procure and maintain livestock damage prevention measures such as electric fences, livestock guarding dogs, or permanent herding, either via the Common Agricultural Policy (CAP) (Marsden and Hovardas, 2020) or similar schemes (e.g. Agridea, 2022). Lethal interventions, which are illegal unless allowed by governments to remove problem individuals (Ordiz et al., 2013), are an additional element of damage prevention, yet its efficiency remains controversial due to a lack of empirical, conclusive evidence (Santiago-Ávila et al., 2020; Bruns et al., 2020). The varying quality and scope of implementation of non-lethal measures has given way to a debate on whether and what kind of damage prevention performs best (Eklund et al., 2017; Bonnet et al., 2019; Oliveira et al., 2021). A large-scale randomized control trial on the effectiveness of different prevention measures is still missing to date (van Eeden et al., 2018).

To shed more light on our understanding of livestock damages caused by wolves, several studies attempted to identify factors explaining their spatial variation. Factors consistently found relate to husbandry practices (e.g. Pimenta et al., 2017; Kaczensky, 1999) with free-ranging livestock most at risk during nights in winter. For other factors, however, reported results mostly appear conflicting. At regional scales, for instance, several studies reported that livestock in heterogeneous landscapes (Kaartinen et al., 2009b) with intermediate agricultural use (Fowler et al., 2019) and particularly in proximity to forest edges (Rigg et al., 2011), were the most vulnerable to wolf predation. Across multiple countries, however, no landscape features were found to correlate with the number of compensated sheep (Gervasi et al., 2021a). In contrast, the number of wolves correlated positively with the number of compensated sheep at the scale of multiple countries (Gervasi et al., 2021a), yet at regional scales, incidents were reported to increase with the geographic spread of wolves (Harper et al., 2005; Khorozyan and Heurich, 2022), but not with an increase in their numbers (Khorozyan and Heurich, 2022).

In an attempt to reconcile these findings, we compiled a large Europe-wide data set of incident-based livestock damage incidents at the municipality level from 2018 to 2020. We then characterized their distribution in space and time and examined the extent to which regional densities in damages can be explained by wolf presence and landscape features reflecting the density and overlap between wolves and livestock. We further estimated regional trends in damage incidents, which we argue are helpful indicators for coexistence of wolves and humans.

2. Material and methods

2.1. Case-based livestock damage incidents

We collected case-based livestock damage data for 2018, 2019 and 2020 at the regional level (i.e. NUTS3 regions, see below), where cases are incidents of livestock depredation as recorded by authorities. While most reported incidents reflect a single attack of wolves on livestock, they may rarely involve multiple attacks if livestock was not checked daily. To obtain case-based data, we consulted the websites of regional authorities if available. Otherwise, we reached out to regional and national authorities of all EU member states, Norway and Switzerland (Supplementary Table S.1) in spring 2019, 2020 and 2021 to report livestock damage incidents of the previous year using a template questionnaire (Supplementary Table S.2). Contacts were mediated by a collaborator from the EU Life EuroLargeCarnivores programme. Our questionnaire consisted of fixed-response questions to be filled per incident, with an option to comment in a separate column. The main attributes were (i) the primary asset missing, injured or killed, (ii) the assessment level or probability of the cause being identified correctly, (iii) the amount of compensation paid per incident, and (iv) the damage prevention measure implemented at the time of the incident in the broad categories defined by Eklund et al. (2017): *electric fence*, *wire fence*, *livestock guarding dog*, *permanent shepherd*, and *other qualified protection.*

We translated the submitted information to English. If the questionnaire was returned incomplete, we followed up and entered any additional information we received by hand. While many respondents adhered to our fixed-answer request, some replies had to be curated manually to match our standards: (i) If more than one asset species was reported for the same incident, we recorded the incident for each species separately, but kept the same incident ID and treated the event as a single incident in our analyses; (ii) if no assessment level on the certainty of wolf predation was reported, we recontacted the authorities for clarification. In case we did not receive any information, we chose the category *unspecified.* (iii) If more than one date was reported for the same incident, we took the first reported date. (iv) If no geographic coordinates were submitted indicating the location of the damage incident, we used the village name (or the smallest geographic unit available) and converted it into geographic coordinates using the Google

Geocoding API (Google, 2022). If neither geographic coordinates nor geographic units were given, or if the provided name could not be converted to coordinates, we removed the incident from our analysis.

As we accumulated data annually, we sent along a report including descriptive statistics as well as the finalized national data set of the previous year for cross-checking by the authorities. In addition, we shared initial exploration of the data to demonstrate the relevance of such data.

2.2. Geographical regions

We conducted our analyses at different geographic scales: at the continental and country levels, as well as at the three levels of the Nomenclature of Territorial Units for Statistics (NUTS; European Commission and Eurostat, 2020) that subdivide each country into smaller geographic units. The NUTS regions mostly follow the administrative subdivisions of the EU Member States (see Fig. 1 for visualization). In

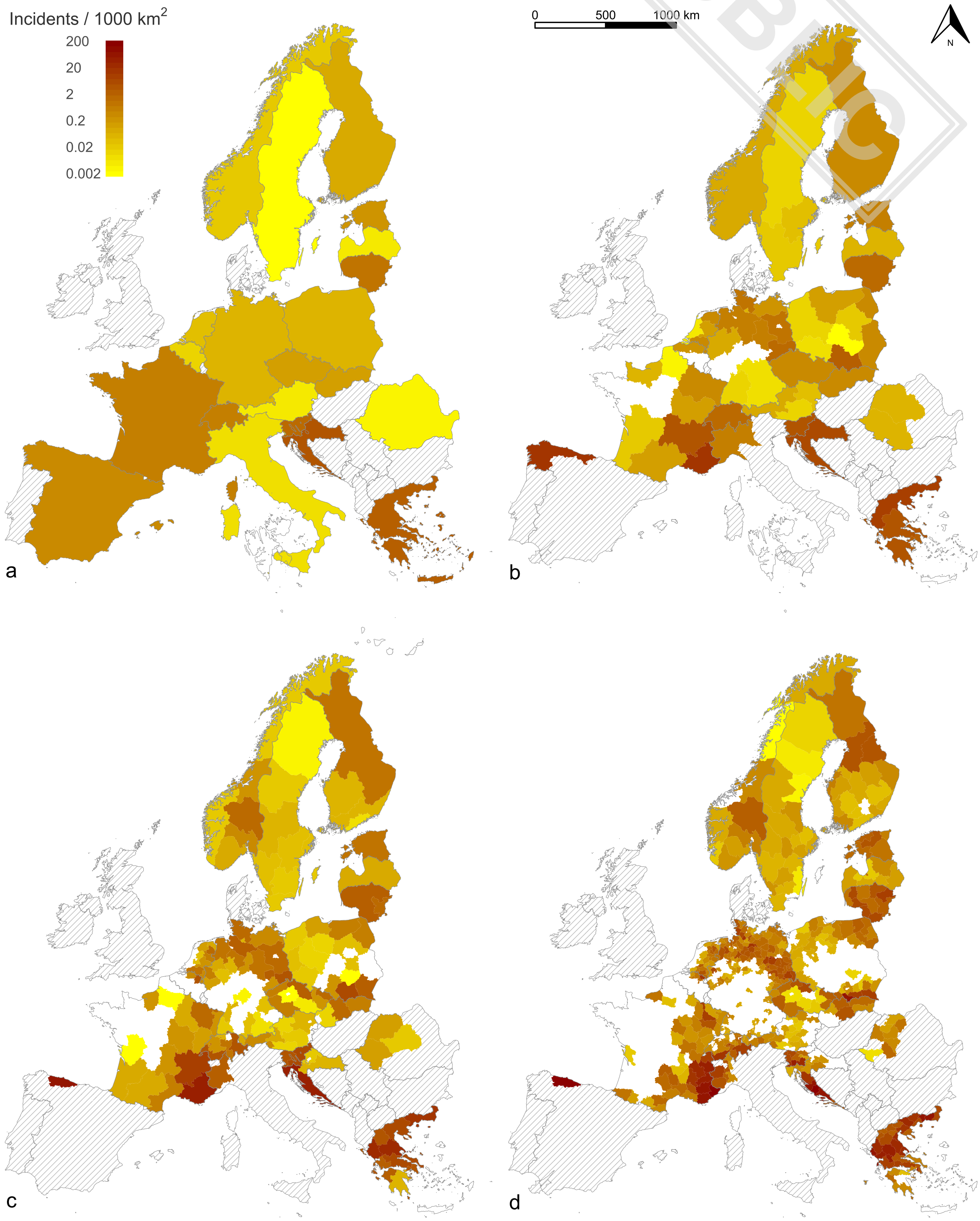

Fig. 1. Wolf-caused livestock incident densities plotted on the country level (a), NUTS1 level (b), NUTS2 level (c) and NUTS3 level (d) across the years 2018, 2019 and 2020. Gray shaded regions indicate regions from which we did not obtain data. Regions shown in white indicate regions from which no damages were reported.

addition, they are unambiguously standardized across Europe and are strictly hierarchical: each country is composed of one or more NUTS1 region, each of which is composed of one or more NUTS2 regions and so forth. Benefiting from this hierarchical setup, we first compiled the counts n_{ik} of reported damage incidents for each year $Y_k \in \{2018, 2019, 2020\}$ and all 470 NUTS3 regions $i = 1, \ldots, I$. Next, we obtained counts $n_{rk} = \sum_{i \in r} n_{ik}$ for all geographic regions r in NUTS2, NUTS1, each country and the continent as a whole by summing across all NUTS3 regions i encompassed in r, denoted here as $i \in r$.

We restricted these counts to incidents for which wolves were sufficiently likely the cause: for administrative regions that provided an assessment level, we kept those with category *presumed correct* or *confirmed*. For administrative regions that did not provide an assessment level, we considered only incidents for which a compensation was paid. If neither was provided, we considered all submitted incidents assuming that only sufficiently probable incidents were shared with us.

2.3. Seasonal distribution of damage incidents

To characterize the seasonal distribution of damage incidents, we aggregated all available incidents by month, discarding all incidents for which no date was provided. To test for temporal variation, we performed χ^2-tests on monthly counts versus their expectation under a uniform distribution. To test if incidents generally occur later in the year in northern than southern Europe, we performed Mann-Whitney-U tests on the months, grouping all incidents reported from Estonia, Finland, Latvia, Lithuania, Norway and Sweden (north) and Croatia, France, Greece, Italy and Spain (south). All tests were performed with the statistical software R (R Core Team, 2021), using the functions chisq.test() and wilcox.test().

2.4. Covariates explaining wolf-caused damage incident counts

2.4.1. Considered covariates

We investigated whether the following ten covariates may explain difference in damage incident counts between NUTS3 regions.

- **Area of wolf presence** To characterize wolf presence across Europe, we used the shapefile compiled for the most recent period available (2012 to 2016) at a 10 × 10 km resolution for the Large Carnivore Initiative of Europe, IUCN Specialist Group and for the IUCN Red List Assessment (Kaczensky et al., 2021). This map encompasses the entire region considered in this study. At each grid point, the authors translated the presence and frequency of wolves into one of three categorical variables: *permanent*, *sporadic* or *no presence*. We overlaid this map with the NUTS3 regions using the st_intersection() function from the *sf* package (Pebesma, 2018) in R (R Core Team, 2021) and used this intersection to determine the area (in km^2) permanently and sporadically occupied by wolves for each NUTS3 region, denoted by *Per-A* and *Spo-A*, respectively.
- **Wolf area by land cover classes** We used the CORINE Land Cover (CLC, code 18) data from the year 2018 (European Environment Agency, 2018) to quantify land cover for all analyzed regions. The available data covers our area of interest at a scale of 1:100,000. The classification comprises artificial surfaces, agricultural areas, forests and semi-natural areas, wetlands and water bodies. We used the st_intersection() function from the *sf* package (Pebesma, 2018) in the statistical software R (R Core Team, 2021) to overlay the CLC layers with the area occupied by wolves (either permanently or sporadically, see above), and further with each NUTS3 region. This way, we obtained the areas of broad-leaved (*BLF-A*), coniferous (*CF-A*) and mixed forests (*MF-A*) occupied by wolves as a proxy of suitable wolf refuge areas, and the areas of pastures (*P-A*) and natural grasslands (*NG-A*) occupied by wolves as a proxy of wolf hunting areas and livestock presence.
- **Historical continuity of wolf presence** Following Gervasi et al. (2021a) and based on previous estimates (Chapron et al., 2014), we determined for each NUTS3 region whether (1) or not (0) wolves were present during the 1950–1970s, a factor denoted by *50ya*.
- **Grazing season length** The mean per region of the bioclimatic variable *BIO11* (mean temperature of coldest quarter) of WorldClim (Fick and Hijmans, 2017), which was previously found to be a good predictor of the grazing season length across Europe, explaining 52 % of the total variation (Phelan et al., 2016).
- **Support for prevention measures** To quantify policies regarding the support of livestock damage prevention schemes, we used an ordinal variable *Prev* stating whether prevention measures were financially supported between 2018 and 2020 in a given region (*yes*, *partially*, or *no*) as defined in Marsden and Hovardas (2020). Our data is related to the purchase of fencing and livestock guarding dogs, as data on their funding is most commonly available. The partial support means that the financial support provided does not cover full costs of the prevention measures. We used information from Marsden and Hovardas (2020) for Croatia, Finland, France, Greece, Latvia and Lithuania. For the remaining regions we used information provided by the European commission (*European Union*, 2022), or (DBBW, 2021; Alpconv, 2020; Naturvardsverket, 2022).

To test whether we missed any major environmental factor, we also explored models that considered for each NUTS3 region i) the latitude of the centroid, ii) the average altitude as provided by the Digital Elevation Model for Europe (dowloaded from https://www.mapsforeurope.org/datasets/euro-dem on November 2, 2022) and iii) the mean of each bioclimatic variable available from WorldClim (BIO1 through BIO19) after transforming all temperatures to Kelvin.

2.4.2. Considered models

Let us denote by $z_{rc}, c = 1, \ldots, C$ and $f_{rd}, d = 1, \ldots, D$ sets of numerical and factor covariates for each region r, respectively. We considered two types of models to account for the heteroscedasticity present in the data:

1. A Poisson model with log-link function

$$n_r \sim \text{Pois}\left(exp\left[\alpha_0 + \sum_c \alpha_c log(1 + z_{rc}) + \sum_d \beta_f f_{rd}\right]\right),$$

where α_0 is an intercept and α_c and β_f the regression coefficients. All numerical covariates (e.g. area occupied by wolves) were log-transformed to maintain their expected linear relationship with incident counts, but we added one to each to allow zero values. We further fitted this model on the sum of annual counts across years $n_r = \sum_k n_{ik}$ to account for the non-independence of annual damage incidents within region regarding the covariates tested, i.e. to ensure they all share the same error term without the need for a hierarchical model. The use of the sum is justified since the sum of Poisson random variables is itself Poisson distributed (Johnson et al., 1997). For the few NUTS3 regions for which we obtained data for two years only, we scaled the sum by 3/2 and rounded it to the nearest integer.

2. To avoid the need for log-transforming covariates and rounding, we also fitted a Gaussian model with power-transform of the form

$$n_r^\tau \sim \alpha_0 + \sum_c \alpha_c z_{rc} + \sum_d \beta_f f_{rd}$$

where τ denotes the parameters of the power transform. We chose the value of τ that maximized the variance explained of the full model using all covariates, and identified it using a line search. As for the Poisson case, the sum of Gaussian random variables is Gaussian itself, justifying the use of n_r (Balakrishnan et al., 2016).

To identify the best sub-models (i.e. selection of covariates), we used the function dredge() from the package *muMin* (Barton and Barton, 2015) in R, and the Akaike Information Criterion (AIC) to identify the best models. We further used the function anova() from the package *stats* (Bates et al., 1992) to determine the fraction of the total variation explained by each model and to test for the significance of included covariates using χ^2 and F tests under the Poisson and Gaussian case, respectively.

2.5. Trends in livestock damages

We estimated trends in livestock damage incidents for each region r using a Bayesian inference approach similar to that in Aebischer et al. (2020) that accounts for survey gaps and stochastic variation, but extended to more than two time points. Let n_{ik} denote the observed incident counts in NUTS3 region i in year Y_k, $k = 1,\ldots,K$. We assumed these counts are Poisson distributed $n_{ik} \sim \text{Poisson}(\lambda_{ik}s_i)$ with means proportional to two region-specific factors: the rate λ_{ik} at which incidents occur in the region i during year Y_k, and rate s_i with which incidents are reported in that region.

We assumed that for any region r, incident rates follow a common exponential trend with the rate γ_r such that $\lambda_{ik} = \lambda_{i0}exp(Y_k\gamma_r)$ for all $i \in r$, and we sought to infer the trend γ_r from all incident counts n_{ik} reported for all NUTS3 regions $i \in r$ for all years k. To do so, we conditioned on the total numbers of counts $\nu_i = \sum_k n_{ik}$ across years for each NUTS3 region (see Link and Sauer, 1997). The conditional distribution of $\boldsymbol{n}_i = (n_{i1},\ldots,n_{iK})$ given the total number of counts ν_i is multinomial:

$$\boldsymbol{n}_i|\nu_i \sim \text{Multinom}(p_{i1},\ldots,p_{iK}),$$

(Johnson et al., 1997) in our case with probabilities

$$p_{ik} = \frac{\lambda_{i0}exp(Y_k\gamma_r)s_i}{\sum_{l=1}^{K}\lambda_{i0}exp(Y_l\gamma_r)s_i} = \frac{exp(Y_k\gamma_r)}{\sum_{l=1}^{K}exp(Y_l\gamma_r)}, \tag{1}$$

where the sum runs across all years $l = 1,\ldots,K$. Due to conditioning, the nuisance parameters λ_{i0} and s_i are canceled out from the fraction, rendering trend estimates independent of any variation in reporting rates across administrative regions.

The likelihood of the full observation vector $\boldsymbol{n} = (\boldsymbol{n}_i, i \in r)$, conditional on $\boldsymbol{\nu} = (\nu_i, i \in r)$, is

$$f(\boldsymbol{n}|\gamma,\boldsymbol{\nu}) \propto \prod_{i \in r,k} p_{ik}^{n_{ik}}.$$

Following Aebischer et al. (2020), we chose the non-informative Jeffrey's prior for γ_r, which is (up to a normalizing constant) the square root of the determinant of the Fisher information

$$\mathcal{I}(\gamma_r) = -\mathbb{E}\left[\frac{d^2}{d\gamma_r^2}logf(\boldsymbol{n}|\gamma_r,\ \boldsymbol{\nu})\right].$$

Using $\mathbb{E}[n_{ik}] = \nu_i p_{ik}$ and $\sum_k p_{ik} = 1$, we arrive at

$$\mathcal{I}(\gamma_r) = \sum_i \nu_i \sum_k \frac{\dot{p}_{ik}^2}{p_{ik}},$$

where

$$\dot{p}_{ik} = \frac{dp_{ik}}{d\gamma_r} = exp(Y_k\gamma_r)\frac{\sum_{l=1}^{K}exp(Y_l\gamma_r)(Y_k - Y_l)}{\left[\sum_{l=1}^{K}exp(Y_l\gamma_r)\right]^2}.$$

We implemented an R package (see data availability statement) to determine the posterior $\mathbb{P}(\gamma_r|\boldsymbol{n}_i,\nu_i)$ using trapezoidal integration of the constant $\mathbb{P}(\boldsymbol{n}_i|\nu_i) = \int \mathbb{P}(\boldsymbol{n}_i|\gamma_r,\nu_i)\mathbb{P}(\gamma_r)d\gamma_r$ and determined the posterior mode using a line-search. Using the function birp_data(), we created one data set per NUTS region for which damage incidents were reported for at least two years (setting all efforts to 1.0) and then inferred trends for this region using the function birp().

We then classified each region as having an increasing or decreasing trend if the posterior mode $\hat{\gamma}_r > 0$ or $\hat{\gamma}_r < 0$ respectively, and quantified the uncertainty p_{γ_r} associated with these point estimates as the posterior probability indicating the opposite sign:

$$p_{\gamma_r} = \begin{cases} \int_{-\infty}^{0} \mathbb{P}(\gamma_r|\boldsymbol{n}_i,\nu_i)d\gamma_r & \text{if } \hat{\gamma}_r > 0 \\ \int_{0}^{\infty} \mathbb{P}(\gamma_r|\boldsymbol{n}_i,\nu_i)d\gamma_r & \text{if } \hat{\gamma}_r < 0 \end{cases}$$

Using the above method, we inferred trends for the total number of incidents combined across all affected species as well as for each species individually. Pearson correlations among the species-specific trend estimates were calculated using the function cor.test in R (R Core Team, 2021), restricting the calculation to regions for which trends could be estimated for both species.

To test if trends were higher in regions only recently colonized, we used a Mann-Whitney U test on the posterior modes against either *50ya* or a factor indicating whether a NUTS3 region was bordering other regions without damage incidents (1) or not (0).

2.6. Testing for spatial autocorrelation

We used Moran's I to test for autocorrelation in the estimated damage trends (posterior modes) and other metrics across each NUTS3 region. For each pair of regions i and j, we used a weight w_{ij} indicating whether the regions share a common border ($w_{ij} = 1$) or not ($w_{ij} = 0$), assessed using the function st_touches() from the package *sf* (Pebesma, 2018) in R (R Core Team, 2021). We assessed the significance of I against a null distribution obtained by permuting the values randomly across regions one million times.

To test for variation on the north-south cline, we further determined the latitude of the centroid of each NUTS3 region using the function st_centroid() of the *sf* package.

2.7. Explaining variation in livestock damage incident trends between regions

For Germany, information on wolf occurrences is available at a finer spatial and temporal scale from (BIJ12 et al., 2022) since 2000. We mapped these occurrences on NUTS regions with st_intersection() as above and calculated the number of known wolf units w_{rk} per NUTS3 region r for the years $Y_k \in \{2018, 2019, 2020\}$. In contrast to previous analyses (e.g. Reinhardt et al., 2019), we treated wolf individuals, pairs and packs each as one territorial unit since single individuals may also cause extensive damage.

We then used the function cor.test() in R to test for correlations between the number of wolf units in each region w_{rk} and the number of reported incidents n_{rk}, limiting the analyses to regions for which wolves, incidents or both were reported. We further used these data to test for correlations between trends in livestock damage incidents and trends in wolf occurrences. For this, we inferred trends in the number of known wolf units $\gamma_r^{(w)}$ for each NUTS region r using the same approach as described above for incidents. We then tested for correlations between the trends inferred for wolf units ($\gamma_r^{(w)}$) and those inferred for damage incidents (γ_r) at all NUTS levels, considering regions for which at least one wolf was reported in any of the years 2018, 2019 or 2020 using Pearson and Spearman correlations in R (R Core Team, 2021). Finally, we tested if the inferred damage trends correlate with the number of years that wolves were present in each region, defined as the number of years between the first reported wolf and 2020.

3. Results

We collected data on livestock damage incidents caused by wolves for 2018, 2019 and 2020 from national or regional authorities for the following 16 countries: Austria, Belgium, Croatia, Czech Republic, Finland, France, Germany, Greece, Latvia, Netherlands, Norway, Poland, Slovakia, Slovenia, Sweden and Switzerland. We obtained partial data for five additional countries: for Estonia we could only obtain data for 2018 and 2019, and for Lithuania only for 2019 and 2020. For Italy, Romania and Spain, we received data only for a subset of the provinces (seven, eight and one NUTS3 regions, respectively). Requests were declined or left unanswered by five countries: Belarus, Denmark, Hungary, Portugal and Ukraine. In total, we obtained data for 910 NUTS3 regions, of which 470 reported incidents. The total number of reported incidents was 43,703, of which 43,513 (99.6 %) could be unambiguously attributed to a single NUTS3 region and were kept for our analyses. These incidents were distributed as 13,895, 15,086 and 14,532 across the three years 2018, 2019 and 2020, respectively.

We further restricted our analyses to incidents for which wolves were sufficiently likely the cause. A subset of countries (Austria, Belgium, Switzerland, Germany, Estonia, Greece, Croatia, Latvia, Netherlands, Norway, Poland, Sweden, Slovenia) provided an assessment level. Of the 22,112 incidents from these countries, we kept 17,904 (80.97 %) that were reported as *confirmed* (4803, 21.7 %) or *presumed correct* (13,101, 59.2 %), and excluded incidents that were *negative* (2017, 9.1 %), *uncertain* (706, 3.2 %), *no assessment possible* (1200, 5.4 %), *assessment pending* (269, 1.2 %) or *unspecified* (16, 0.1 %). For Finland, Romania and Spain that did not provide an assessment level, we kept the 9181 (99.5 %) incidents for which compensation was paid. For the remaining countries (Czech Republic, France, Italy, Lithuania, Slovakia) that provided neither information, we kept all 12,177 incidents, assuming that only sufficiently probable incidents were shared with us. In total, we thus kept 39,262 (89.8 %) incidents (Supplementary Table S.3) and will refer to these as *wolf-caused incidents* below.

The countries with the highest numbers of reported wolf-caused incidents across the three years were France (9840), Greece (6870) and Spain (6856). The countries with the lowest numbers of reported wolf-caused incidents were Belgium (79), Latvia (91) and Austria (115). As shown in Fig. 1, regions varied greatly in their densities of wolf-caused incidents, with south-eastern France, coastal Croatia, northern Greece and the Spanish province of Asturias being regional hotspots of livestock damage incidents in our data set.

Most data collected was not associated with the information on the application of damage prevention measures (84.3 %), with only eight countries (Belgium, Croatia, Germany, Latvia, Netherlands, Poland, Slovenia, Sweden) reporting whether or not a prevention measure was applied at the time of the incident. Among those incidents (6158, 15.7 %), the most common measure was *electric fence* (767 incidents, 12 %), followed by *wire fence* (311, 5 %), *guarding dog* (31, 0.5 %) and *permanent shepherd* (8, 0.13 %). For an additional 3826 (62 %) of incidents the prevention measure was indicated as *other qualified protection*, while 1430 (23 %) affected unprotected animals. Note that for 224 incidents (3.6 %), multiple measures were in place.

3.1. The species most frequently targeted by wolves

At the continental scale and in terms of wolf-caused incidents, sheep were most frequently affected (21,301, 54.2 %), followed by cattle (7672, 19.5 %) and goats (4328, 11 %). Other animals less frequently affected included horses (3125 wolf-caused incidents, 8 %), reindeer (1976, 5 %), dogs (529, 1.4 %), domestic deer (red, roe or fallow deer, 201, 0.5 %), donkeys (166, 0.4 %), pigs (10, <0.1 %) and lamas or alpaca (8, <0.1 %). For 343 (0.9 %) additional incidents, the affected animals were not indicated to the species level. For Finland, the most affected species was reindeer (85.8 %), for Greece cattle (46.5 %) and for Spain (Asturias) horses (42.3 %). For the remaining 18 countries, the most frequently affected species was sheep (46.0–97.6 %).

Across the 39,262 wolf-caused incidents reported from all 21 countries, 99,056 animals were killed, injured or went missing. The distribution of the number of affected animals per incident was heavily right-skewed with 58.9 % of all incidents involving a single animal, 23.2 % involving two or three and only 3.5 % involving ten or more individuals. Only two incidents involved more than 100 animals, the largest being the only reported incident from the Romanian province of Timis affecting 402 animals. Sheep had the most causalities (71,023, 71.7 %), and goats had more casualties (11,338, 11.4 %) than cattle (8415, 8.5 %) in line with goat incidents usually involving more animals (2.6 on average, standard deviation 2.6) than cattle incidents (1.1 on average, standard deviation 0.6). Incidents involving sheep involved 3.3 animals on average (standard deviation 5.2).

3.2. Seasonal distribution of wolf-caused incidents

To gain insights into seasonal patterns of livestock damage incidences, we aggregated records by months, discarding 359 (0.9 %) incidents which did not provide a date. As shown in Fig. 3, incidents showed strong temporal variation ($\chi^2 = 4480.9$, $p < 10^{-15}$). Across all species, incidents peaked between July and October with 48.7 % of the total incidents falling within these months. This pattern was particularly visible for sheep (55.2 %), as well as for cattle (43.4 %) and goats (40.9 %), albeit less pronounced. In contrast, incidents involving horses peaked between April and July (51.8 %) and those involving reindeer between September and December (67.5 %).

The differences between target species are explained by the geographic distribution of livestock and the observation that incidents in northern Europe (Estonia, Finland, Latvia, Lithuania, Norway and Sweden) generally occured later in the year than in southern Europe (Croatia, France, Greece, Italy (Piemonte) and Spain (Asturias), $U = 51,011,510$, $p < 10^{-15}$): in southern Europe, 23.3 % of all incidents occured before May, but in northern Europe only 5.1 %. This pattern was also found for each species with enough data to perform the Mann-Whitney-*U* test, namely sheep ($U = 14,902,386$, $p < 0.042$), cattle ($U = 678,569$, $p < 10^{-8}$), goats ($U = 87,644$, $p < 0.001$), horses ($U = 1,411$, $p < 0.001$) and dogs ($U = 22,569$, $p < 10^{-4}$).

3.3. Covariates explaining wolf-caused damage incidents

The livestock species affected by wolf depredation vary greatly across Europe, largely due to climatic factors and local husbandry practices (see above). To gain more insight into general factors explaining the variation in wolf-caused incidents, we thus focused on the combined incidents across all species.

Across the NUTS3 regions, only 2.1 % of the total variation in the number of wolf-caused incidents was within regions across years, while 97.9 % was across regions ($F = 93.8$, $p < 10^{-15}$). To explain that latter part, we conducted regression analyses on the total number of wolf-caused incidents across years within each NUTS3 region. As explanatory covariates we used 1) the total area occupied by wolves, either sporadically (*Spo-A*) or permanently (*Per-A*), 2) the area within regions occupied by wolves for the CORINE land cover classes Pastures (2.3.1, referred to as *P-A*), Broad-leaved forests (3.1.1, *BLF-A*), Coniferous forest (3.1.2, *CF-A*), Mixed forests (3.1.3, *MF-A*) and Natural grasslands (3.2.1, *NG-A*), 3) whether or not wolves were present at their lowest extent during the 1950–1970s (*50ya*), 4) the degree of governmental support for prevention measures (*Prev*) and 5) the mean temperature of the coldest quarter (*BIO11*) as a predictor for grazing season length.

We first used Poisson models with log-link function and log-transformed covariates to explain incident counts. All covariates have explanatory power, as evidenced through an ANOVA performed models including a single covariate which were all significant ($p < 10^{-16}$ for all covariates, Fig. 2). The best model according to the Akaike Information Criterion (AIC) (Supplementary Table S.4) included all ten covariates, explained $R^2 = 63.2\%$ of the total variation, and was significantly better than the next-best sub-model with fewer covariates ($\Delta AIC = 12.76$, Burnham and Anderson, 2004). For that model, all included covariates were also significant under an ANOVA ($p < 0.003$ for all covariates).

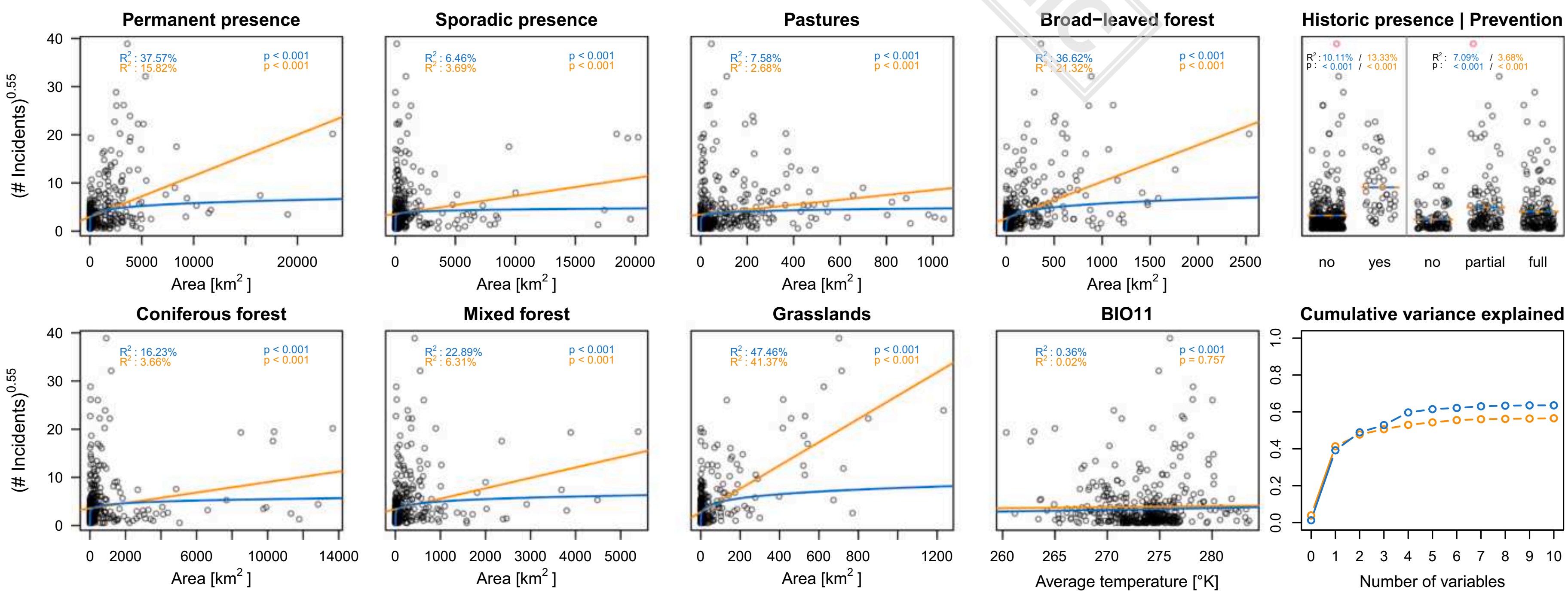

Fig. 2. Correlation of ten environmental covariates and the number of reported wolf-caused livestock damage incidents in our data set, excluding the outlier region of Asturias (ES120). The Poisson model is shown in blue, the Gaussian model is shown in orange. For visualization purposes, both models are plotted using the same power-transformation on the incidents. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Nonetheless, the majority of the variance appeared to be explained by a rather small set of covariates (Fig. 2). By itself, *NG-A*, for instance, explained $R^2 = 47.5\%$, which was more than 3/4 of the total variance explained by all covariates. Other covariates with high explanatory power included *Per-A* ($R^2 = 37.6\%$), *BLF-A* ($R^2 = 36.6\%$), *MF-A* ($R^2 = 22.9\%$), *CF-A* ($R^2 = 16.2\%$) and *50ya* ($R^2 = 10.1\%$), while the remaining had $R^2 < 10\%$ and *BIO11* even only $R^2 = 0.4\%$. Allowing for two covariates, the best model included *NG-A* and *Per-A* and explained $R^2 = 57.1\%$ of the total variance, or $> 90\%$ of the variance explained using all ten covariates. The best models using three covariates further included *BLF-A* and explained $> 95\%$ of the variance explained by all ten covariates, while five covariates were sufficient to explain $> 99\%$. Notably, a model just including *Per-A* and *Spo-A* explained $R^2 = 37.7\%$ and thus significantly less than *NG-A* alone ($\Delta \text{AIC} = 3243.18$).

We obtained qualitatively similar results when using a Gaussian model on the raw covariates with power-transformed incident counts, for which we estimated the best transformation exponent to be $\tau = 0.54$. In this case, all covariates except *BIO11* had explanatory power on their own (ANOVA, $p < 10^{-16}$ for all covariates, Fig. 2) and a model including all covariates had $R^2 = 56.6\%$. The best model according to AIC explained $R^2 = 56.0\%$ using the seven covariates *Per-A*, *Spo-A*, *NG-A*, *CF-A*, *P-A*, *BIO11* and *50ya* (Supplementary Table S.4). There were, however, several models not significantly different from the best model ($\Delta \text{AIC} < 2.0$), including a model using all covariates except *Prev*, and under the best model all covariates were significant when tested using an ANOVA ($p < 0.0001$) except *P-A* ($p = 0.64$). Again, *NG-A* had the biggest individual explanatory power with $R^2 = 41.3\%$, matched 3/4 of the total variance explained by the best model and explained significantly more than the $R^2 = 16.1$ of a model including *Per-A* and *Spo-A* ($\Delta \text{AIC} = 154.82$). The covariates with the next highest explanatory power, however, differed in order from the Poisson model and included *BLF-A* ($R^2 = 21.4\%$), *Per-A* ($R^2 = 15.9\%$) and *50ya* ($R^2 = 13.4\%$). The remaining had $R^2 < 10\%$, again with *BIO11* explaining the least ($R^2 < 0.1\%$).

To avoid spurious fitting, the results above were obtained after excluding the outlier region of Asturias, Spain, which had more than twice as many incidents than the next region and thus contributed disproportionate to the total variance. When including this region, however, results were qualitatively similar: *NG-A* explained the most variance under both a Poisson and Gaussian model, the best model with two covariates included additionally *BLF-A* in both cases, and the best model included all ten covariates in the Poisson case but fewer covariates in the Gaussian case (Supplementary Fig. S.1, Supplementary Table S.5).

Residuals of the best models were significantly spatially autocorrelated ($I = 13,554.8$ and $I = 843.2$ under the Poisson and Gaussian models without outlier, $p < 10^{-6}$ in both cases), suggesting that some additional variance may be explained with landscape features or other spatial factors not included in our model. To test if our choice of covariates was lacking any additional major environmental effect readily available for all NUTS3 regions, we extended our models with altitude, latitude and all bioclimatic variables available from WorldClim. Under the Poisson model (without outlier), adding all these 20 additional covariates explained an additional 9.2% of the total variation ($R^2 = 72.4\%$). However, this is likely a result of over-fitting and difficult to interpret: when added to the base model of ten covariates, the most informative additional covariate (*BIO9*) explained a mere extra 1.3% of the total variation, and all others an extra 0.4% or less. Similar results were obtained under the Gaussian model, where the most informative covariate (also *BIO9*) explained an extra 0.3% only.

In contrast, several interaction terms among the ten chosen covariates appear meaningful. Under the Poisson model, 43 of the 45 possible interaction terms led to significantly better models ($\Delta \text{AIC} > 2.0$) when added individually to the model containing all ten considered covariates. Of those, two explained more than an extra 4% of the total variation: *50ya* × *Spo-A* ($R^2 = 68.0\%, \beta = 0.44$) and *50ya* × *NG-A* ($R^2 = 67.5\%, \beta = -0.43$). Under the Gaussian model, 24 of the 45 possible interaction terms led to significantly better models, of which five explained an extra 4% or more of the total variation: *NG-A* × *P-A* ($R^2 = 66.0\%, \beta = -0.00007$), *NG-A* × *BIO11* ($R^2 = 62.2\%, \beta = 0.0028$), *NG-A* × *Prev* ($R^2 = 61.7\%, \beta = 0.0016$), *NG-A* × *50ya* ($R^2 = 61.0\%, \beta = -0.0200$) and *BLF-A* × *50ya* ($R^2 = 61.0\%, \beta = -0.0099$).

We finally quantified the benefit of restricting landscape features to areas occupied by wolves and found that for the Poisson and Gaussian cases respectively, the best models using landscape features of the entire NUTS3 regions explained 2.6 % and 10.0 % less of the total variance than the best models presented above.

3.4. Number of wolf units correlated with incidents

For Germany, detailed information is available on the number of wolf units in each NUTS region for each of the three years studied here. Focusing on the NUTS3 regions for which livestock damage incidents, wolves or both were reported, the number of wolf units was significantly correlated with the number of reported incidents for each year ($p < 0.001$, $p < 0.001$ and $p = 0.002$, respectively) as well as for all years combined ($p < 0.001$). The magnitude of the correlation diminished over time, from $\rho = 0.60$ in 2018, to $\rho = 0.49$ and $\rho = 0.39$ in 2019 and 2020, respectively.

The number of wolf units across the three years was a slightly worse predictor of the number of reported incidents per region within Germany than the *Per-A* and *Spo-A* derived of a distribution map for 2012–2016 ($R^2 = 0.37$ vs. $R^2 = 0.38$ for the Poisson model and $R^2 = 0.37$ vs. $R^2 = 0.44$ for the Gaussian model).

3.5. Trend analysis

We estimated trends of wolf-caused incidents across the three years 2018, 2019 and 2020 for all geographic regions with at least one incident reported from at least two years, accounting for survey gaps and stochastic variation. At the continental scale, our analysis indicated with certainty that incidents were increasing ($\mathbb{P}(\gamma > 0|\boldsymbol{n}) = 1.0$) with an estimated rate of $\widehat{\gamma} = 0.021$ per year (posterior mode), translating into an 4.2 % increase from 2018 to 2020. At smaller geographic scales (Fig. 4), the pattern is rather heterogeneous: of the 320 NUTS3 regions with sufficient data (two years with damage incidents), we estimated a positive trend ($\widehat{\gamma}_r > 0$) for 195 (61 %) and a negative trend ($\widehat{\gamma}_r < 0$) for 125 (39 %), with posterior modes spanning from $\widehat{\gamma}_r = -2.91$ for NO074 (Troms og Finnmark, Norway) to $\widehat{\gamma}_r = 3.41$ for FRC13 (Saône-et-Loire, France). Despite this heterogeneity, the $\widehat{\gamma}_r$ estimates, and thus the directionality of the trends, were spatially autocorrelated ($p < 0.003$), but not correlated with the recency of colonization, neither when using the historical wolf distribution *50ya* ($U = 6250$, $p = 0.75$) nor when comparing regions in the center of the wolf distribution (all neighboring regions had damages) to those at the frontier (bordering regions without damages, $U = 1347$, $p = 0.24$).

We also estimated trends individually for the most commonly affected species (sheep, goats, cattle and horses, Supplementary Figs. S.2-S.5). Trends did not appear to be correlated between any pair of species at NUTS1, NUTS2 or NUTS3 level ($p > 0.06$ in all cases), likely because trends could be estimated only for a partially overlapping subset of regions for each species due to the lower number of incidents and a restricted geographic distribution of some species.

For Germany, we also estimated trends in the number of wolf units for each NUTS3 region with at least one wolf unit reported across the three years 2018, 2019 and 2020. The total number of wolf units across all such regions (157, 186, 203) revealed a rapidly growing population ($\mathbb{P}\left(\gamma_r^{(w)} > 0|\boldsymbol{w}\right) = 0.980$) with $\widehat{\gamma}_r^{(w)} = 0.124$ (posterior mode), corresponding to the growth rate of 13.2 % per year. To confirm that growth rates decreased over time, we estimated them for all three-year intervals from 2006 to 2020 for which at least two NUTS3 regions had wolves in three years. Estimated annual growth rates (posterior modes) decreased from 65.8 % for 2006–2008 to 33.6 %, 32.2 %, 30.9 % and finally 13.2 % for 2018–2020.

There was considerable regional variation, with 47 NUTS3 regions showing positive $(\gamma_r^{(w)} > 0)$ and 26 negative $(\gamma_r^{(w)} > 0)$ trends of wolf units for 2018–2020. We tested whether these trends predict trends in wolf-caused incidents, but did not find such a correlation at any NUTS level ($p > 0.15$ for Spearman correlations and $p > 0.08$ for Pearson correlations in all cases). The trends in wolf-caused incidents did not correlate with the time since wolves were first reported in a region ($p > 0.16$ at any NUTS level). Across Germany, however, wolf units $(\hat{\gamma}_r^{(w)} = 0.124)$ and damage incidents $(\hat{\gamma}_r = 0.077)$ did not grow at significantly different rates $(\mathbb{P}\left(\gamma_r < \gamma_r^{(w)}\right) = 0.732)$.

4. Discussion

We consolidated a large, incident-based data set on livestock damages caused by wolves across Europe in recent years. A total of 16 countries reported complete data for the years 2018, 2019 and 2020, and additional five countries reported partial data. The majority of reported incidents involved a single livestock head and only very few involved more than ten individuals. In line with previous reports (Kaczensky, 1999; Bautista et al., 2019; Gervasi et al., 2021a), sheep were the most affected species, both in terms of incidents and affected individuals. There was, however, spatial variation reflecting the regional importance of different livestock species such as reindeer in Finland or horses in Spain (Asturias). Interestingly, however, cattle suffered disproportionately many incidents in Greece, although Greece had the least amount of cattle per sheep in 2019 (according to EuroStat, Table apro-mt_ls) of all EU member states for which we received damage data.

We found considerable seasonal variation with incidents peaking in August and September. There was a clear north-south cline with a much smaller fraction of incidents reported during winter in northern compared to southern Europe, likely because in the north, livestock is kept indoors more often during these months. Finland showed a particular interesting seasonal pattern in the number of reported reindeer incidents, which are much higher in early than in late winter (Fig. 3). This is largely due to the migration of wolves into Finnish reindeer husbandry areas in autumn, leading to many reported incidents, followed by their legal hunting later in winter, which reduces their numbers and consequently reported damage incidents (H. Norberg, personal communication, November 22, 2022).

4.1. Extensively used habitats favor wolf-caused damage incidents

The data also revealed substantial spatial variation in the number of incidents caused by wolves across Europe. A large part of that variation (>40%) is explained by a single environmental covariate: the area of natural grasslands of each region occupied by wolves. Up to about 60 % of the total variation is explained by just a few additional covariates such as the area permanently occupied by wolves or the area of broad-leaved forest occupied by wolves, though their ranking varied with model choices. Regardless of these choices, however, a model only including the area permanently or sporadically occupied by wolves had a much poorer fit. Thus, areas with high numbers of incidents are not only qualified by the presence of wolves, but also through extensively cultivated habitats where wolves may be numerous and livestock more accessible or more difficult to protect (Rigg et al., 2011; Jedrzejewski et al., 2004; Treves et al., 2004; Fowler et al., 2019). Our model thus allows to identify both current and future hotspots where the conflict between wolves and livestock owners may be particularly intense.

Correlations with similar landscape features were recently reported

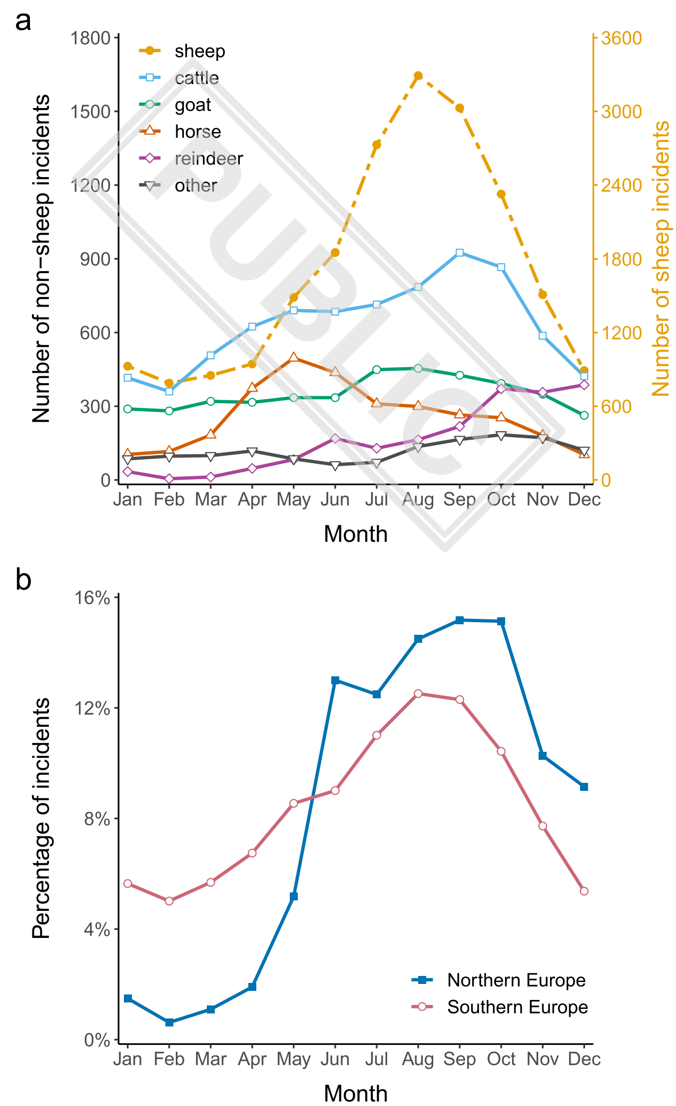

Fig. 3. (a) Monthly distribution of wolf-caused livestock damage incidents. (b) Percentage of monthly wolf-caused livestock damage incidents for northern Europe (Estonia, Finland, Latvia, Lithuania, Norway, Sweden) and southern Europe (Croatia, France, Greece, Italy (Piemonte), Spain (Asturias)).

from damages for Poland (Fedyń et al., 2022), but not from a recent, multi-national analysis on sheep incidents for which compensation was paid (Gervasi et al., 2021a). Based on that data, Gervasi et al. (2021a) reported an effect of historical continuity of wolf presence, but unlike our study they found no correlation with any of the environmental variables tested. This apparent difference is likely due to three reasons. First, our data set encompassed many more NUTS3 regions (470 vs. 140) and hence had more statistical power to detect such correlations. Second, Gervasi et al. (2021a) modeled yearly damage counts for each region through year- and region-specific random effects. We decided against such a strategy and rather focused on the total counts per region, because annual count data are not strictly independent due to, for instance, local husbandry practices and the same wolf individuals causing damage across multiple years. While region-specific random effects may capture that effect, they effectively deprive the model of identifying fixed effects. Third, we considered landscape features only within the areas predicted to be occupied by wolves, rather than in NUTS3 regions as a whole. Indeed, most NUTS3 regions are only partially occupied by wolves, potentially due to historical or political reasons not connected to the landscape features of the entire region. Consequently, the best models using the landscape features of the entire

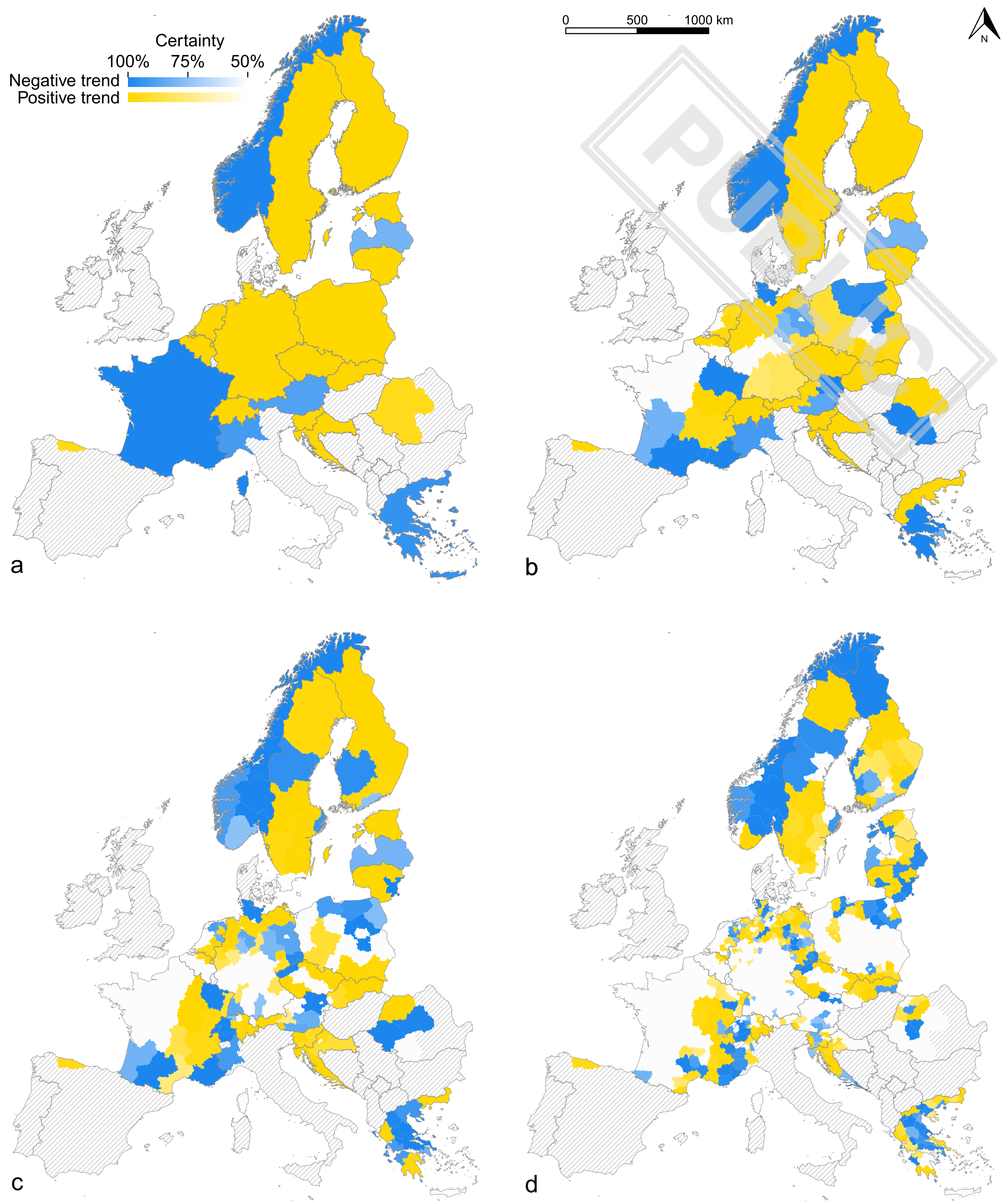

Fig. 4. Wolf-caused livestock damage trends at the country (a), NUTS1 (b), NUTS2 (c) and NUTS3 (d) level. Each region was classified as having an increasing (yellow) or decreasing (blue) incident trend, depending on the posterior mode $\gamma_r < 0$ for decreasing trends and $\gamma_r > 0$ for increasing trends. Colour saturation indicates uncertainty quantified as the posterior mass within the classified interval, ranging from solid (1.0) to white (≤ 0.5). Regions without reported incidents are shown as white, regions with no data reported are shaded in gray. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

NUTS3 regions explained considerably less of the total variation than when using landscape features of areas occupied by wolves. On top of such land cover metrics, the historical continuity of wolf presence then appeared to have a significant, but limited explanatory power.

While our models captured a large part of the between-region variation on wolf-caused damage incidents, about 40 % of it remained unexplained even with our best fitting model. Part of that variance may be due to variation in reporting rates between regions, rather than variation in damage incidents themselves (Gervasi et al., 2021a). Also, incident numbers may be inherently stochastic, especially if wolf numbers are low such that a single problem individual can cause a spike in incidents for a short time. In the NUTS2 region AT12 (Niederösterreich, Austria), for instance, most of the 19 incidents in 2018 could be attributed to a single female wolf. This individual was no longer active in 2019 and 2020, when incidents dropped to two and only one, respectively. In our data set, however, only 2.1 % of the total variation was across years.

Since residuals appeared to be geographically clustered such that too many (or too few) incidents were often predicted for multiple neighboring regions, there likely exist additional covariates predictive of incident numbers. Possible such covariates include the densities of wild prey species (Meriggi and Lovari, 1996), with wolves being more likely to switch to livestock in areas of low prey densities (Kaartinen et al., 2009a), yet also being particularly attracted to areas with high prey densities (Treves et al., 2004), increasing damage incidents in both. While estimates of wild prey densities are currently not available at the continental scale, they may themselves be explained through bioclimatic and land-use variables. When adding all bioclimatic variables of WorldClim as well as altitude and latitude, an extra almost 10 % of the variation in incidents counts could indeed be explained, but none of those additional covariates appeared to explain sufficient extra variation to warrant their discussion.

Interactions among covariates had larger individual contributions, but given their large number, meaningful testing was difficult. It is nonetheless interesting to note that many of the interactions that explained several extra percentages of the total variation involved the historical continuity of wolves, mostly with negative coefficients, suggesting that the main environmental covariates appear to underestimate wolf-caused incidents for regions only recently occupied by wolves. This is in good agreement with the thought that incidents often spike in areas recently colonized by wolves (Trouwborst, 2018; Marucco and Boitani, 2012; Dalmasso et al., 2011; van Eeden et al., 2018; Gervasi et al., 2021a), due to measures preventing livestock damages having been abandoned in the absence of large carnivores or the unfamiliarity of wolves with local conditions and preferential preying on livestock as easy prey (Kaczensky et al., 2021; Linnell et al., 1996).

While the historical continuity of wolf presence since the 1950–1970s is unlikely to capture this effect in full, only limited data is available to test for more recent effects. At the continental scale, for instance, the most recent estimate of wolf presence is dated to the period of 2012–2016 (Kaczensky et al., 2021). This information is already potentially outdated for the frontier of the ongoing wolf recolonization in Europe as we received reports of wolf-caused incidents for 78 NUTS3 regions presumably without any wolves present. The very low number of annual incidents reported from these regions (median of 1.0), however, remained well explained by zero-value area covariates. In Germany, for which more detailed and up-to-date data on the presence of wolves is available and for which 23 NUTS3 presumed wolf-free regions reported incidents, the wolf area covariates derived from the outdated distribution maps turned out to be even better predictors of the number of incidents than the actual number of wolf units present.

While wolves were not present in most regions within Germany about ten years ago, they have since made a successful comeback (BIJ12 et al., 2022). While Reinhardt et al. (2019) reported an annual growth rate of 36 % for the period 2000–2015, it appears that the growth has been slowing down steadily to 14 % for the three years considered here. Along with wolves, livestock damage incidents have increased at comparable rates at the larger scale, but the connection between the number of wolves and the number of reported incidents appears complex at local scales as we did not find any correlation between trends in the number of wolf units and trends in the number of damage incidents on the NUTS3 level. Thus, a locally growing wolf population does not seem to imply a growing number of incidents, a finding previously reported for sheep lost to wolves in Germany (Khorozyan and Heurich, 2022). Second, while the number of wolves in a region seems to correlate well with the number of damage incidents, this correlation appears to diminish over time, in line with a lack of correlation between these trends. Thus, with wolves establishing themselves in more regions at higher numbers, the relationship between wolves and damage incidents becomes more obscure and complex.

4.2. *Lack of data on the effectiveness of prevention measures*

A likely interpretation of these findings is that a resumed presence of wolves may lead to a more widespread adoption of protective measures that reduces wolf-caused damage over time, or at least that the relationship between damage incidents and wolf presence is modulated by variation in the adoption of such measures. However, the dataset gathered here is not ideal to directly test such hypotheses, nor hypotheses about the effectiveness of different prevention measures, as we lack information on their use. To assess the effectiveness of damage prevention measures, the frequency of damage incidents should be contrasted between paired protected and unprotected sites before and after the applications of measures. The authorities, however, tend to only record the prevention measure in place at the time of an incident, if at all, which does not allow conclusions on their effectiveness. In our dataset, for instance, the most frequently reported prevention measure was *electric fence*, most likely not because it was ineffective, but because it was very commonly used.

Despite the large amount of public money spent on supporting prevention measures, a large-scale randomized controlled trial on the effectiveness of different prevention measures is still crucially missing. As an alternative, the data reported here could be overlaid with georeferenced data on the use of prevention measures and the accurate data on the presence of wolves, provided that the spatial autocorrelation in the use of prevention measures can be accounted for. To the best of our knowledge, such data is currently not available at a large geographic scale. We thus strongly encourage authorities to collect and share such information in the future, ideally using specific categories that distinguish among the diverse set of measures used, rather than grouping them into the broad categories used here. We note, however, that unbiased collection of such information may prove difficult if livestock owners fear consequences for improperly implemented measures.

4.3. *Trends in damage incidents help inform damage management*

Across the years 2018–2020 studied here, about 60 % of all NUTS3 regions studied showed an increasing trend in damage incidents, reflecting the growth of the European wolf population and the need for conflict mitigation. In Germany, for which we have detailed information on the distribution of wolves, for instance, the number of wolf units and damage incidents grew at comparable rates. While we lack information on the growth of wolf populations at the continental scale, regional trends in damage incidents illustrate the small-scale nature of livestock damage incidents and their mitigation. First, we estimated negative trends for 39 % of all NUTS3 regions, although wolf populations unlikely shrank in any of these (Chapron et al., 2014). Second, and while the inferred trends were spatially auto-correlated with incidents either increasing or decreasing in multiple neighboring regions, the trends did not point uniformly in one direction for any country, neither at NUTS3 or NUTS2 levels. Third, and in contrast to a recent study in Italy (Gervasi et al., 2021b), we did not find trends to systematically differ between regions with established wolf populations compared to those more recently colonized. Fourth, the trends inferred individually for each of the most commonly affected livestock species were not correlated.

Some of the above variation may be arise from random fluctuations across the just three years studied here. In addition, the lack of correlation across affected species may be partly due to a limited number of incidents per species and a limited geographic overlap in their use. Of the 272 NUTS3 regions for which we could infer the trends for the most commonly affected species (sheep), only for 119 NUTS3 regions (43.8 %) trends could also be inferred for the second most commonly affected species (cattle). This overlap is further reduced to 5 NUTS3 regions (1.8 %) when comparing sheep and reindeer.

Nonetheless, local trends in damage incidents provide particularly useful indicators of the intensity of the conflict between wolves and livestock owners. In contrast to analyses on raw incident counts (Gervasi et al., 2021a), trend estimates are comparable at any geographic level as they are robust to variation in sampling effort and data collection across regions (Link and Sauer, 1997; Aebischer et al., 2020). Together with raw incidents counts, they provide local stakeholders with direct feedback about their mitigation efforts.

With wolf populations in Europe generally growing and expanding their ranges, a decreasing trend indicates a reduction in the local conflict. If such trends are observed after a specific management intervention such as the adoption of additional prevention measures, other changes in husbandry practices, or the successful management of problematic individual wolves, they provide direct feedback on its success. That seems particularly helpful in regions with currently high numbers of incidents, where decreasing trends may serve as a legitimation for the action taken and proof that mitigation is possible, even if incident numbers may currently be far from acceptable levels. Conversely, nearly stable or even increasing trends in regions with high numbers of incidents suggest that current damage prevention and mitigation measures seem insufficient and/or ineffective, thus requiring additional attention to prevent the escalation of the conflict, particularly in the case of increasing trends. In regions with currently low numbers of incidents, decreasing or nearly stable trends serve as a confirmation of the current policies, while increasing trends act as an early warning sign that extra effort needs to be taken to maintain incidents at current levels, for instance because the local wolf population is growing. We therefore invite regional and national authorities to continue integrating their damage data into accessible data bases, and to use trends over longer time periods, at larger geographic scales, or both, to effectively monitor and help mitigate human-carnivore conflicts across Europe.

CRediT authorship contribution statement

Liam Singer: Methodology, Software, Formal analysis, Writing – original draft, Writing – review & editing, Visualization, Project administration. **Xenia Wietlisbach:** Formal analysis, Project administration, Visualization, Writing - original draft, Writing - review & editing. **Raffael Hickisch:** Conceptualization. **Eva Maria Schoell:** Formal analysis, Writing – review & editing. **Christoph Leuenberger:** Methodology. **Angela Van den Broek:** Investigation, Writing – review & editing. **Manon Désalme:** Investigation, Writing – review & editing. **Koen Driesen:** Investigation. **Mari Lyly:** Investigation, Writing – review & editing. **Francesca Marucco:** Investigation. **Miroslav Kutal:** Investigation, Writing – review & editing. **Nives Pagon:** Investigation, Writing – review & editing. **Cristian Remus Papp:** Investigation, Writing – review & editing. **Paraskevi Milioni:** Investigation, Writing – review & editing. **Remigijus Uzdras:** Investigation. **Ilgvars Zihmanis:** Investigation. **Fridolin Zimmermann:** Investigation, Writing – review & editing. **Katrina Marsden:** Writing – review & editing. **Klaus Hackländer:** Investigation. **José Vicente López-Bao:** Investigation, Writing – review & editing. **Sybille Klenzendorf:** Investigation, Writing – review & editing. **Daniel Wegmann:** Conceptualization, Methodology, Software, Writing – original draft, Writing – review & editing, Supervision, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The data that supports the finding of this study, namely the full records of damage incidents as well as all environmental covariates used to model them, are openly available at Zenodo (doi:https://doi.org/10.5281/zenodo.7797522). Note that some information considered sensitive by the authorities was withheld (compensation payments and/or exact geolocation data), but can be obtained by contacting the authorities directly (Supplementary Table S.1). The above Zenodo link further includes all R code written to conduct our analyses, with the exception of the R package to perform the trend analyses, which is available at https://bitbucket.org/wegmannlab/birp/src/master/.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biocon.2023.110039.

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RESEARCH ARTICLE

Conflict Misleads Large Carnivore Management and Conservation: Brown Bears and Wolves in Spain

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Abstract

Large carnivores inhabiting human-dominated landscapes often interact with people and their properties, leading to conflict scenarios that can mislead carnivore management and, ultimately, jeopardize conservation. In northwest Spain, brown bears *Ursus arctos* are strictly protected, whereas sympatric wolves *Canis lupus* are subject to lethal control. We explored ecological, economic and societal components of conflict scenarios involving large carnivores and damages to human properties. We analyzed the relation between complaints of depredations by bears and wolves on beehives and livestock, respectively, and bear and wolf abundance, livestock heads, number of culled wolves, amount of paid compensations, and media coverage. We also evaluated the efficiency of wolf culling to reduce depredations on livestock. Bear damages to beehives correlated positively to the number of female bears with cubs of the year. Complaints of wolf predation on livestock were unrelated to livestock numbers; instead, they correlated positively to the number of wild ungulates harvested during the previous season, the number of wolf packs, and to wolves culled during the previous season. Compensations for wolf complaints were fivefold higher than for bears, but media coverage of wolf damages was thirtyfold higher. Media coverage of wolf damages was unrelated to the actual costs of wolf damages, but the amount of news correlated positively to wolf culling. However, wolf culling was followed by an increase in compensated damages. Our results show that culling of the wolf population failed in its goal of reducing damages, and suggest that management decisions are at least partly mediated by press coverage. We suggest that our results provide insight to similar scenarios, where several species of large carnivores share the landscape with humans, and management may be reactive to perceived conflicts.

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Introduction

Many populations of large carnivores are threatened, usually due to anthropogenic causes [1,2,3]. This is often due to loss of habitat and high mortality levels related to depredation, other damages to properties, competition for game species, or threat to humans (e.g. [4]). On the other hand, the ongoing increase of some large carnivore populations in Europe and North America raises concern of increasing wildlife-related conflicts, as broadly defined by a confrontation between people with different views, e.g. those supporting protection and functional carnivore conservation vs. those supporters of intensive management [5,6].

Few studies on damages caused by large carnivores have actually explored the ecological, economic and societal correlates that lay behind such conflict scenarios [5,7]. However, subjective components (i.e. cultural, emotional) are important to understand and eventually mitigate wildlife-related conflicts, which may substantially affect wildlife management and conservation [8]. Furthermore, when two or more large carnivore species are sympatric, the mixture between objective (ecological, economic) and subjective components may lead to particularly complex diagnosis, as one species may suffer disproportionate negative human attitudes, unrelated to the actual magnitude of damages [7,9]. Such context calls for sound evaluation of the factors involved in conflict scenarios and the outcome of management actions [10].

Lethal population control, i.e., culling, is actually a main tool to manage large carnivores in conflict scenarios [11], implicitly assuming that carnivore abundance is a key driver of the amount of damages. Conflict scenarios related to brown bears *Ursus arctos* and wolves *Canis lupus* are common in Europe [12,13], and our study area in the Cantabrian Mountains of NW Spain is no exception [14,15]. The area holds sympatric populations of brown bears and wolves in the south-western edge of their European distributions, and both are isolated and distant from other bear and wolf populations [16]. While brown bears in Spain are listed as "critically endangered" and fully protected (about 200 individuals in the Cantabrian Mountains [17]), wolves are considered "near threatened" (about 250 packs in Spain, about 70 in the Cantabrian range [18]). Wolves are a game species in most of their Spanish range, and are also subject to regular culling. Management of bears and wolves in our study area includes economic compensations for damages. In addition, management of wolves includes annual culling programs, allegedly assuming that culling mitigates depredation on livestock and conflict.

We used records of damages to human properties and their press coverage to analyze a conflict scenario with two large carnivore species subject to distinct management. We explored correlates between damages and ecological (i.e. abundance of predators, harvested wolves, livestock numbers, harvested ungulates), monetary (economic cost of compensations) and societal (media coverage) variables. In addition, we discuss whether annual wolf culling programs followed legal mandates, and succeeded in preventing damages and reducing conflict.

Methods

We analyzed records of complaints on depredation on beehives and livestock by bears and wolves, respectively, in the autonomous region of Asturias, NW Spain (10,604 km^2; Fig 1). Asturias holds about 80% of the Cantabrian brown bear population [17], and about 30 packs of wolves. It is the only region in Spain that pays for damages by bears and wolves in its entire territory as part of recovery and management plans, respectively. Asturias is also the only Spanish administration that has detailed datasets of damages caused by both species. We compiled available data on wolf and bear abundance, complaints on damages by both species and details of damages, compensations paid to those complaints, livestock numbers, harvested ungulates and number of wolves killed in culling programs; all these data were provided by the regional administration with management responsibilities for both species.

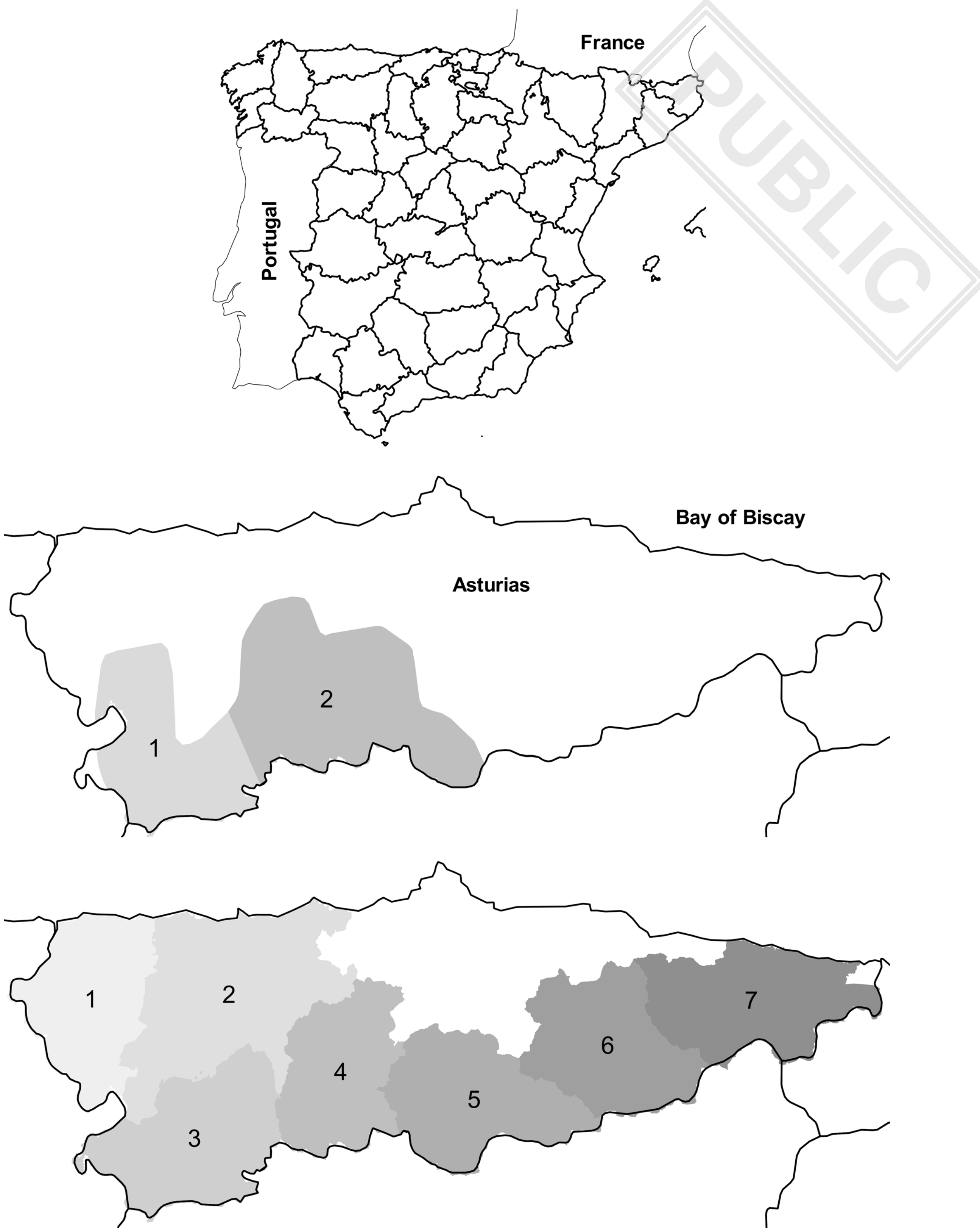

Fig 1. Study area. Top panel: Asturias autonomous region (NW Spain, shaded). Intermediate and bottom panels: brown bear and wolf study zones, respectively, in Asturias. Basemaps made with Natural Earth, public domain map data available at http://www.naturalearthdata.com/.

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Table 1. Variables used in the study.

Variables	Description	Period (N years)
beehives	Response: beehives damaged by bears per year	1991–2008 (18)
depredation	Response: livestock heads depredated by wolves per year	2003–2010 (8)
Fcub	Female bears with cubs of the year in the current year	1991–2008 (18)
$Fcub_{-1}$	Female bears with cubs of the year in the previous year	1990–2007 (18)
packs	Wolf packs in the current year	2003–2010 (8)
culled	Wolves culled in the current year	2003–2010 (8)
$culled_{-1}$	Wolves culled in the previous year	2002–2009 (8)
$ungulates_{-1}$[a]	Ungulates shot in the previous year	2003–2010 (8)
livestock[b]	Livestock heads (× 10^3) per year in wolf zones	2003–2010 (8)
compensations	Annual cost of damages (€ × 10^3) by bears and wolves	2003–2010 (8)
news	Annual news on damages by bears and wolves	2004–2010 (7)

[a] Roe deer, red deer, wild boar and chamois hunted per year.

[b] Sheep, goats, cattle and horses.

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Compensations of damages by bears and wolves are paid after verification by rangers in the field. Files included the number of affected beehives or livestock heads, and the amount paid as compensation in each case. Data availability was not consistent for all variables and periods; hence we used slightly different periods in the various analyses (Table 1 and S1 Dataset).

Bear and wolf data

Bear management in Asturias follows a mandated recovery plan (Decree 9/2002 [19]). We used annual counts of females with cubs of the year, the only available metric of bear abundance in our study area, as a demographic surrogate for the bear population; numbers of female bears with cubs of the year were available since 1982 [20]. We differentiated two zones to analyse bear data based on well differentiated food resources [21] (Fig 1).

Bear use of anthropogenic food sources may increase when natural resources are scarce and / or when bear abundance is higher. To evaluate the latter hypothesis we used the number of damaged beehives as response variable, and the number of female bears with cubs (during any given year and in the previous one) and year as potentially explanatory variables. Claims of bear damages included beehives, livestock, orchards, and various other damages to properties. We chose the number of damaged beehives as response variable because beehives comprised 85% of damage claims to both beehives and livestock during the studied period, and 70% of monetary paid compensations; in addition, they are more robustly reported through the administrative process. The lack of reliable records on the number of beehives in Asturias prevented estimation of the proportion of beehives affected by bear attacks.

Wolf management in Asturias followed a mandated management plan during our study period (Decree 155/2002 [22]). It includes annual culling quotas of wolves based on three criteria: a) wolf abundance, b) trend and amount of damages, and c) level of social conflict. We used the official, available data on annual numbers of wolf packs, wolves killed in culling programs, attacked livestock heads and paid compensations Counts of packs were the only available annual metric of wolf abundance. There was no data available on the level of "social conflict", or any description of its precise meaning. Data were provided by the Asturian government, the administration responsible of the wolf management plan in the whole territory of Asturias. Wolf management is divided into 7 zones; we followed a similar scheme to analyze damages on livestock (Fig 1).

Wolves in the Cantabrian Mountains prey on wild ungulates (roe deer *Capreolus capreolus*, wild boar *Sus scrofa*, red deer *Cervus elaphus* and chamois *Rupicapra parva*) and on livestock [23]. We hypothesized that livestock heads compensated for attacks by wolves per management zone and year would be positively correlated with the number of wolf packs, the number of ungulates harvested the previous year, and livestock numbers. Conversely, it would be negatively correlated with the number of wolves culled in the previous year. Data on free-ranging livestock in Asturias are publically available and updated annually [24]. Data on wild ungulates harvested by hunters per season was also provided by the regional administration.

Media coverage of bear and wolf damages

We used media coverage as proxy of the perception of risk associated to large carnivores. Our approach is based on conceptual framework on risk judgement by the general public [25, 26], which has also been applied to perceptions of wildlife risk in mass media [27, 28]. We hypothesized that the number of damage-related news for bears and wolves would be proportional to the cost of compensations (€) paid for damages.

We searched for news on wolf and bear damages in 2004–2010 in the digital archive of the only newspaper that covers all the region of Asturias (*La Nueva España*, LNE; www.lne.es). LNE had an estimated readership of 351,000 daily readers in 2010 [29], about one third of the population of Asturias. In addition, it has three daily sub-regional editions, covering the central, eastern and western areas of the region.

To collect and classify news about damages by both species, we followed a procedure similar to [30]. Specifically, we searched for strings "oso" (bear) and "lobo" (wolf) in the digital archive of LNE. For each entry, we read first the headline of the story, which usually allowed us discarding unrelated uses of the terms (e.g. movies, surnames, etc.). Then we checked secondary headlines to allow coding stories as damages to beehives or attacks to livestock, searching also for the string "daños" (Spanish for damages, a term widely used in this context). Thereby we discriminated damage news from any other news about bears and wolves. We finally assigned each story to the municipality where it applied, and to zones in the case of wolves.

Lethal control of wolves and management criteria

We sought to determine if the number of wolves legally killed every year in each zone was related to wolf management criteria: a) the number of wolf packs present per year and zone; b) compensations paid (€) for verified damages per year and zone; and c) the number of damage-related news per year and zone, as a proxy to conflict. The analysis of media coverage of wolf damages per zone was restricted to 2006–2009, when media archives allowed assigning news to specific zones.

Data analysis

First, we analysed if there were trends in the variables (exponential growth rate), fitting generalized linear models (GLM; Poisson distribution) with year as explanatory variable. Then we fitted generalized mixed models (GLMMs with negative binomial distribution, logit link function) [31] to damages, with zone as random factor. We evaluated model performance and parsimony using Akaike Information Criteria (AIC), the difference (ΔAIC) between each candidate model and the best model (lowest AIC), and AIC weights (AICw [32]). Analyses were performed in R and SAS [33, 34].

Results

In the study area there were 8 ± 3 female bears with cubs per year (mean ± SD). Bears damaged 250 ± 237 beehives annually, and the cost of bear damages averaged 127,203 ± 39,779 € per year. The three variables increased over the study period (Table 2). News on bear damages amounted to just 3 ± 1.3 per year (mean ± SD), preventing trend analysis. Beehives damaged by bears in any given year and zone were positively related to the number of bear females with cubs in the previous year (Table 3).

In the study area and period there were 29 ± 5 wolf packs per year (mean ± SD). 15 ± 7 wolves per year were killed in culling programs. The annual number of livestock heads affected by wolf damages averaged 2,951 ± 478, and increased during the study period (Table 2). Compensation costs of wolf damages averaged annually 691,498 ± 201,687 €, and also increased during the study period (Table 2). Livestock heads compensated by depredations amounted to 0.69 ± 0.14% of free-ranging livestock, which averaged 423,079 ± 29,136 heads per year in the study area.

Livestock depredation in any given year and zone was positively related to wolf packs and the number of wolves culled both during the current and the previous year (Table 3; Fig 2). The second and third best models also retained a positive effect of the number of ungulates harvested in the previous year (Table 3). 70% of compensated livestock heads (N = 13,194) were lost between April and October. 7,976 ± 1,011 wild ungulates were shot per year in the study area.

Overall, media coverage on wolves and bears was similar (125 ± 32 and 116 ± 29 news per year, respectively; mean ± SD). The cost per complaint averaged 339 € for wolves and 505 € for bears, although total compensations paid were five times higher for wolves than for bears. The total number of news on wolf damages was 30 times higher than news on bear damages. Media coverage on wolf damages per zone was also uncorrelated to the economic cost of damages (Kendall's tau correlation coefficient = 0.17; N = 35; five years, seven zones).

Most wolves were killed between January and August (71%; N = 101), i.e. including the wolf breeding season. The annual number of wolves culled in each zone ranged from 0 to 11, with an average of 2 individuals per zone and year. Wolf culling was positively related to the number of news on wolf damages per zone, and to paid compensations (Table 4; Fig 3). The number of packs per zone (average = 4; range 1–8) was also retained in the second best model (Table 4).

Table 2. Trends in the variables used in the study.

Variables	EGR[a] (± SE)	P
beehives	0.19 ± 0.03	< 0.001
depredation	0.05 ± 0.01	< 0.001
Fcub	0.06 ± 0.01	< 0.001
packs	0.01 ± 0.03	NS
culled	0.03 ± 0.06	NS
ungulates	0.04 ± 0.01	< 0.001
livestock	-0.02 ± 0.01	< 0.001
compensations (bears)	0.09 ± 0.03	0.01
compensations (wolves)	0.10 ± 0.01	< 0.001
news (bears)	0.05 ± 0.11	NS
news (wolves)	-0.12 ± 0.02	<0.001

[a] Annual trend of each variable estimated as exponential growth rate (± SE) via GLMs with Poisson distribution.

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Table 3. Models fitted to beehives damaged by bears, and to livestock heads depredated by wolves.

beehives[b]	AIC	ΔAIC	AICw	ß ± SE[a]	P
null model	411.5	17.3	0		
Fcub + Fcub$_{-1}$ + year	395.7	1.5	0.32		
Fcub$_{-1}$ + year	394.2	0	0.68		
Variables retained					
Fcub$_{-1}$				0.27 ± 0.12	0.03
year				0.14 ± 0.04	0.002
depredation[c]					
null model	733.1	25	0		
packs+culled+culled$_{-1}$ +ungulates$_{-1}$ +livestock	711.1	3	0.13		
packs +culled +culled$_{-1}$ +ungulates$_{-1}$	709.5	1.4	0.29		
packs +culled +culled$_{-1}$	708.1	0	0.58		
Variables retained					
packs				0.06 ± 0.03	0.08
culled				0.09 ± 0.02	0.001
culled$_{-1}$				0.07 ± 0.02	0.001

GLMM models with negative binomial distribution and zone as random factor. AIC is Akaike Information Criterion; ΔAIC is the difference between best model (lowest AIC) and each candidate model; AIC_w are AIC weights.

[a] Estimate and standard error for the variables retained in the best models.

[b]N = 36; 18 years, two zones.

[c]N = 56; 8 years, 7 zones.

Variables: Fcub, number of bear females with cubs of the year; Fcub$_{-1}$, number of bear females with cubs of the year in the previous year; packs, number of wolf packs in the current year; culled, number of wolves killed in the current year; culled$_{-1}$, number of wolves killed in the previous year; ungulates$_{-1}$, number of ungulates shot in the previous year; livestock, heads of livestock present in the current year.

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Discussion

Conflict scenarios rooted in human attitudes and confronting perceptions of large carnivores, e.g. groups that oppose carnivore recovery vs. carnivore supporters, are major obstacles for carnivore conservation and recovery [35]. Therefore, disentangling the relative importance of ecological, economic and societal factors involved in human-carnivore interactions should facilitate coexistence [36]. We used the number of news on wolf damages per zone as a proxy of social conflict, and found that the press coverage of wolf damages was not correlated to their economic costs. The unbalanced press coverage is relevant because news stories on damages correlated to wolves killed in management actions (Fig 3; Table 4). Media coverage is thus a potential driver of public risk perception of large carnivores (e.g. [26, 28]), showing that conflict resolution does not necessarily lay just on ecological grounds [37], or in science communication. Indeed, social factors may influence management actions (e.g. Fig 3).

We found that livestock damages were positively correlated to wolf culling intensity in the previous year, hinting an undesired outcome of management based on culling. The relation between wolf culling and subsequent damages corresponded to a set of paired years and wolf zones (Fig 2; Table 3); it did not depend on overall trends in wolf numbers or damages, but actually showed a relation between culling and the number of damages the year after. Previous studies showed that culling or hunting do not necessarily minimize depredation on livestock [38,39] and recent research in North America even found similar counter-expected effects in black bears, pumas, and wolves [40,41,42]. To our knowledge, a positive correlation between number of culled large carnivores and increased damages has never been published in Eurasia.

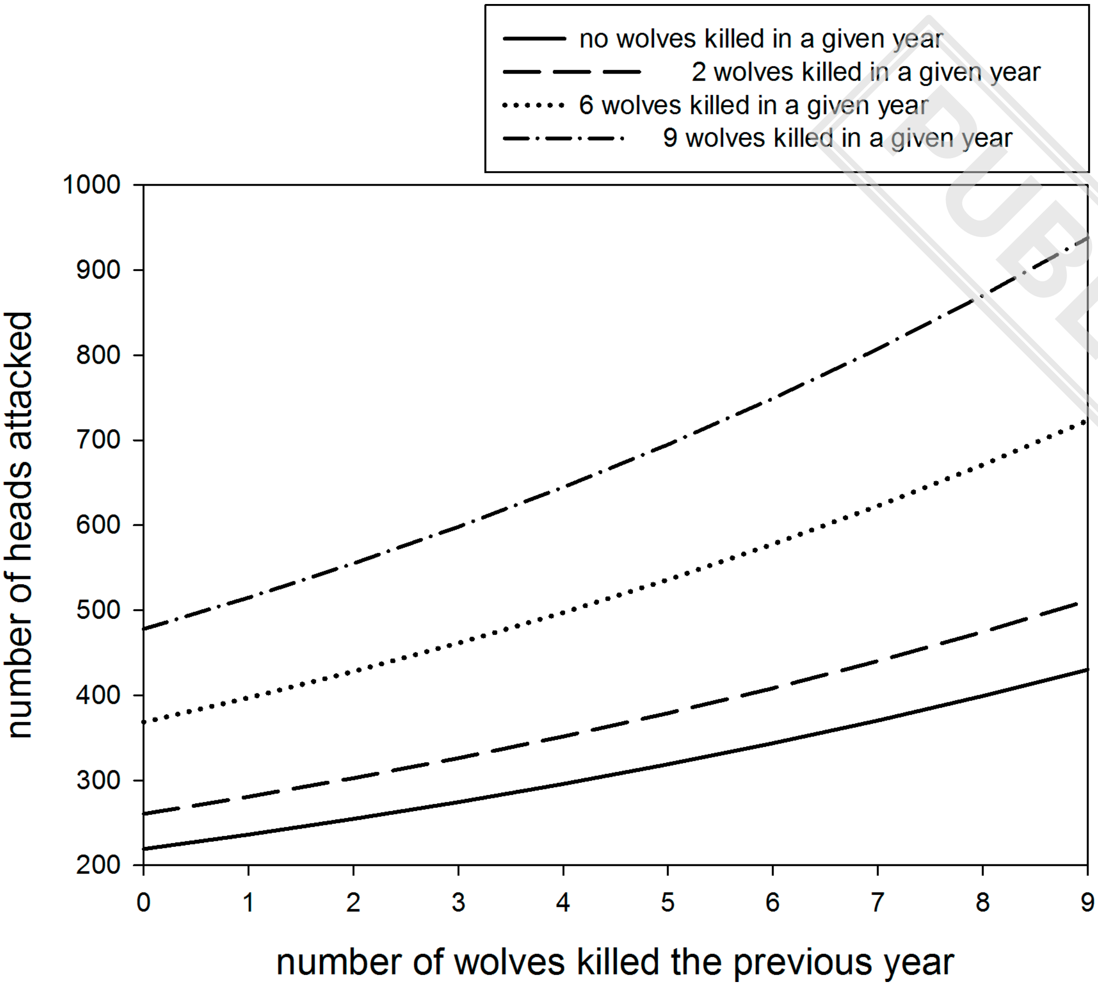

Fig 2. Relationship between the number of livestock heads depredated by wolves and number of wolves culled the previous year. The plot is based on the best model of wolf depredation on livestock; the model was parameterized for different numbers of wolves killed in the current year, and in a zone harboring the average number of packs per zone (N = 4).

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Several plausible scenarios could explain those effects: source-sink hypothesis (e.g. [41]), and social disruption, i.e., an outcome of random culling in highly social animals like wolves [43]. Culling reduces pack size, which together with the social disruption caused by killing

Table 4. Models fitted to the number of wolves culled per year.

	AIC	ΔAIC	AICw	ß ± SE[a]	P
null	110.7	9.7	0		
packs + compensations + news	102.7	1.7	0.30		
compensations + news	101	0	0.70		
Variables retained					
compensations				0.001 ± 0.0002	0.006
news				0.053 ± 0.018	0.008

GLMM models with negative binomial distribution and zone as random factor; N = 28 (four years, seven zones). AIC is Akaike Information Criterion; ΔAIC is the difference between best model (lowest AIC) and each candidate model; AIC_w are AIC weights.

[a] Estimate and standard error for the variables retained in the best model. Variables: packs, number of wolf packs; compensations: cost of complaints due to livestock depredation by wolves (€); news: number of news published on livestock damages by wolves.

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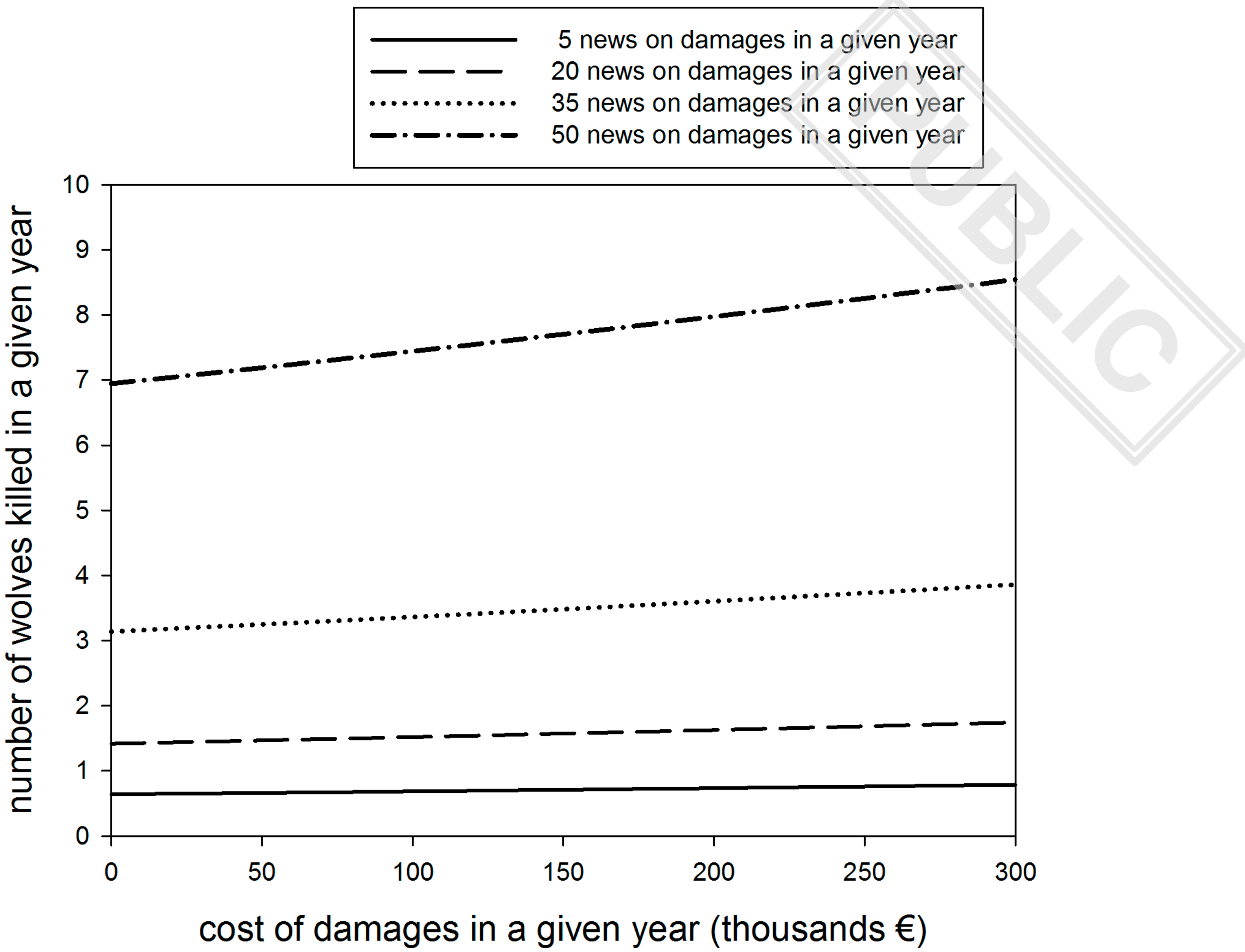

Fig 3. Relationship between wolves culled and compensated damages. The plot is based on the best model relating wolves culled in a given year and the cost of damages compensated in that year, as a function of the number of news on damages published in that year.

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reproductive individuals could result in an increase of the number of packs in a region [44, 45]. In addition, kill rates in wolves depend on season, pack size, prey size and prey density, among others [46, 47]. Kill rates seem to be higher in Europe than in North America, perhaps indicating that higher risk of human-related mortality in European wolves leads to a decline in consumption of each carcass [47, 48]. Although the levels of damages on livestock in our study area may seem disparate for the number of packs and average pack size [49], the observed pattern could arise if wolves spent less time at kills because livestock owners and rangers visit the carcasses. A similar effect has been described for pumas living closer to human residential areas [50].

Availability of wild prey is also an important factor behind carnivore predation on livestock [51, 52]; abundant wild prey may avert predation on livestock. However, data are rarely available to test that idea [53]. We did not have robust data on abundance of wild prey, but our surrogate (ungulates harvested in the previous season) showed a positive correlation with the number of damages by wolves on livestock. Furthermore, unguarded livestock is susceptible to depredation even if wild prey is available [54], adding a human-dependent issue to predator-prey interactions. Livestock husbandry is an objective component that plays a major role in the magnitude of damages by large carnivores [55, 56]. Yet, hard data on type and dedication of husbandry practices are absent in our study area.

The number of bears in the Cantabrian Mountains increased during the study period, coinciding with an increase in damages to beehives. A simple explanation would be that bears shift to anthropogenic resources when the natural ones are scarce, thus increasing damages to human properties. However, we found that bear damages correlated with females with cubs in the previous year. This may indicate that an increase in the proportion of juvenile bears in the population–which have faster growth rates and are often less wary—lead to an increase in damages to beehives.

Bear damages did not seem as conflictive to the press as wolf damages, judging from the dramatic skew in the treatment of damages by bears and wolves: compensations paid annually for wolf damages were indeed five times higher than those paid for bear damages (691,498 v. 127,203 € per year), yet media coverage of wolf damages was 30 times larger (91 v. 3 news per year). Such bias and its potential effects on management can remain undetected when studying only one of several sympatric species in a conflict scenario [12, 57].

Management and conservation implications

A widespread measure to increase social acceptance of large carnivores is to compensate economically the damages they caused [11, 58]. In our study area, about 85% of the complaints were compensated after verification, but compensations did not seem to ease conflict. It is worth noting that stockbreeding activities are subsidized by the Common Agricultural Policy (CAP) of the European Union. Those subsidies are higher for livestock grazing in protected areas, to compensate restrictions associated to them, including potential inconveniences of sharing the landscape with large carnivores and wild ungulates [59, 60].

The situation we described urges the implementation of better livestock husbandry practices instead of wolf culling, which is counterproductive from damage-management and conservation perspectives. Indeed, improving livestock handling is often regarded as the most rational and conservation-oriented measure in different scenarios. It also calls for attention to the role of media and opinion makers as potential amplifiers or drivers of wildlife-related conflicts: wolf depredation affected annually 0.69 ± 0.14% of free-ranging livestock in our study area, i.e., depredation is not a major cause of livestock mortality, but media is seemingly driving the implementation of culling programs.

Culling of populations of apex predators is unjustified on scientific grounds [61]; indeed, culling suppress certain 'apex' traits [62, 63], thus altering their role in ecosystems. In addition, the implementation and outcome of conflict-related management actions on large carnivores should also be evaluated on ethical grounds [45, 64].

Supporting Information

S1 Dataset. Data on damages by bears and wolves used in the analyses. Data on bear and wolf damages, numbers of female bears with cubs, wolf packs, wolves killed in culling programs, harvested wild ungulates, and news on wolf damages used in the analyses of this study. See Table 1 for description of variables.
(XLS)

Acknowledgments

Records of damages by bears and wolves in Asturias, number of female bear with cubs, number of wolf packs, wolves killed in culling programs, and hunting records of wild ungulates, were provided by the Asturian administration, which during our study period was in charge of

implementing the bear management plan in the range of bear distribution (Decree 9/2002, article 4), and the wolf management plan in the whole territory of Asturias (Decree 155/2002, article 5).

Author Contributions

Conceived and designed the experiments: AFG JN ER MD. Performed the experiments: AFG JN AO MQ ER MD. Analyzed the data: AFG JN AO MQ ER. Contributed reagents/materials/analysis tools: AFG JN MQ ER. Wrote the paper: AFG JN AO MQ ER MD.

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RESEARCH ARTICLE

Killing wolves to prevent predation on livestock may protect one farm but harm neighbors

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Data Availability Statement: Certain privacy issues arise from sharing our data. By agreement with the Little River Band of Ottawa Indians, we try to protect the privacy of the livestock owners involved in depredation events. Therefore, we have redacted the precise locational information of complaints and interventions. Hence, the data being included with this submission does not include the precise locations (section information) of each data point. The complete data, with precise location information, can be obtained by written request to: W. Frank Beaver, Director, Natural Resources Department, Little River Band of Ottawa

Abstract

Large carnivores, such as gray wolves, *Canis lupus*, are difficult to protect in mixed-use landscapes because some people perceive them as dangerous and because they sometimes threaten human property and safety. Governments may respond by killing carnivores in an effort to prevent repeated conflicts or threats, although the functional effectiveness of lethal methods has long been questioned. We evaluated two methods of government intervention following independent events of verified wolf predation on domestic animals (depredation) in the Upper Peninsula of Michigan, USA between 1998–2014, at three spatial scales. We evaluated two intervention methods using log-rank tests and conditional Cox recurrent event, gap time models based on retrospective analyses of the following quasi-experimental treatments: (1) selective killing of wolves by trapping near sites of verified depredation, and (2) advice to owners and haphazard use of non-lethal methods without wolf-killing. The government did not randomly assign treatments and used a pseudo-control (no removal of wolves was not a true control), but the federal permission to intervene lethally was granted and rescinded independent of events on the ground. Hazard ratios suggest lethal intervention was associated with an insignificant 27% lower risk of recurrence of events at trapping sites, but offset by an insignificant 22% increase in risk of recurrence at sites up to 5.42 km distant in the same year, compared to the non-lethal treatment. Our results do not support the hypothesis that Michigan's use of lethal intervention after wolf depredations was effective for reducing the future risk of recurrence in the vicinities of trapping sites. Examining only the sites of intervention is incomplete because neighbors near trapping sites may suffer the recurrence of depredations. We propose two new hypotheses for perceived effectiveness of lethal methods: (a) killing predators may be perceived as effective because of the benefits to a small minority of farmers, and (b) if neighbors experience side-effects of lethal intervention such as displaced depredations, they may perceive the problem growing and then demand more lethal intervention rather than detecting problems spreading from the first trapping site. Ethical wildlife management guided by the "best scientific and commercial data available" would suggest suspending the standard method of trapping wolves in favor of non-lethal methods (livestock guarding dogs or fladry) that have been proven effective in preventing livestock losses in Michigan and elsewhere.

Indians, fbeaver@lrboi-nsn.gov, 231-398-2191. Requests should describe the rationale for the data request, as well as what steps will be taken to ensure the privacy of livestock owners involved in depredations remains protected.

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Introduction

Large carnivores, such as gray wolves, *Canis lupus*, are difficult to protect in mixed-use landscapes because some people perceive them as dangerous and because they sometimes threaten human property and safety. Traditionally, governments kill wild animals in an effort to prevent threats to property and safety [1]. However, a recent summary of peer-reviewed studies that employed experimental or quasi-experimental tests of interventions against carnivore attacks on domestic animals in farms raised doubts about the functional effectiveness of lethal methods [2]. Namely, most tests of lethal methods showed no effect or counter-productive effects (higher livestock losses after intervention), and numerous tests contained biases or flaws that preclude reliable inference [2]. Two tests using quasi-experimental designs showed minimal, regional effect of various lethal methods [3] and a strong, local effect of government trapping and aerial shooting [4], respectively. But none provided the highest standard of evidence [2], which are random-assignment experimental tests of an intervention without bias in sampling treatment, measurement, or reporting [5, 6]. Higher standards of evidence were applied to tests of non-lethal methods generally, and two such tests applied the highest standards that also proved effective in preventing predation events on domestic animals (depredation). The two methods were fladry (a visual deterrent effective against wolves only, thus far) and livestock-guarding dogs [7, 8]. A recent controversy over killing wolves in the Northern Rocky Mountains (NRM) illustrates the difficulty of forming scientific consensus on the effectiveness of lethal methods for preventing depredations when standards of evidence are not consistent.

Two teams [4, 9] came to opposite conclusions when analyzing very similar data from the same region and similar period for the Northern Rocky Mountain wolf population. A deeper look suggests that inferences drawn from these quasi-experimental tests are weakened by uncontrolled variables (Box 1).

Box 1

One test included only wolf-killing by aerial gunning and several ground-based methods from 1989–2008 [4], whereas the other included all permitted wolf-killing, including public hunting, from 1987–2012 [9]. The latter of these two analyses found that killing more wolves was followed by more livestock losses the following year, using a negative binomial regression model controlling for multiple variables [9]. However, that test did not account adequately for the time series underlying several variables that increased over time. For example, over time the wolf population increased in size and also spread geographically, thereby exposing more farm animals to depredations. Because the amount of wolf-killing increased over time as (a) recolonizing wolves left the protection of a national park and wild areas, and (b) policy changes introduced wolf-hunting in addition to killing by government agents [4, 10, 11], we should expect the predictors (wolf-killing, livestock exposed, and wolf distribution) to rise over time in parallel with the observed rise in domestic animal losses over time, which would make a statistically significant association spurious if the time trend were not accounted for properly. Another team conducted the same analysis with the same data while accounting for time series trends and statistical misspecifications, and results suggest killing wolves instead led to an increase in attacks on cattle in the same year and fewer attacks the following year, relative to no killing [12]. However, this analysis seems to have eliminated the possibility of an underlying effect of wolf population size and did not consider the geographic spread of wolves, an approach that remains to be validated [12]. Proper statistical control for exposure (encounters between wolves and domestic animals) might require a measure of geographic spread of wolves, not just wolf and domestic animal abundances regionally. The remedy would have required spatial information at scales below that of the region. The authors of the analysis of wolf-killing between 1989–2008 incorporated spatial information, yet did not extend spatial analyses sufficiently, and limited their data to a time period when only government wolf-killing was legally allowed [4]. They found a reduction in risk of recurrence subsequent to wolf-killing within a wolf pack territory. The reductions appeared significant and high in magnitude after an entire pack was killed, and appeared significant but lower in magnitude when only part of a pack was killed, compared with no removal [4]. The analysis was restricted to the affected wolf pack territory, despite the researchers' own work documenting how partial removal of wolves could scatter survivors beyond their original pack range [11, 13]. Therefore, the analysis of risk of recurrence of depredations should have examined neighboring areas and even more distant consequences. The importance of examining livestock loss beyond the edges of wolf pack territories had been noted [14]. We examine the analysis of [4] in greater detail in the Discussion.

We tested the hypothesis that two treatments (lethal and non-lethal intervention) following verified depredations had different effects on the risk of a recurrence (occurrence of a subsequent depredation) at that site and at neighboring sites at two larger geographic scales. We tested that hypothesis because the common justification for lethal interventions worldwide is that eliminating problem individuals, or regional predator reductions, will delay or curtail future losses immediately, and for at least one year until wolves are replaced [15]. We retrospectively examined data collected by state and federal agents in the state of Michigan, USA, from 1998–2014, using methods similar to [4], with two main differences. The first difference was that we examined spatial scales beyond the site of the intervention, so we could detect spill-over effects up to a radius of 16.25 km from the site of the intervention (neighborhood of township scale; see Methods section below). The second difference was that we included 2 distinct interventions: lethal and non-lethal interventions (pseudo-control, see below). Our analysis was retrospective and treatments had not been assigned randomly, thus the highest standard one might achieve would be a silver-standard experiment [2]. With data on the history and locations of events and interventions, we were able to draw stronger inference than a simple comparison of means between interventions. But quasi-experimental tests might be confounded by the effect of time passing (before-and-after) as carnivores, livestock, and people respond to changing conditions and other aspects of the environment change independently.

We had to consider potential bias in treatment. Field agents apparently made subjective judgments about where to implement lethal intervention when that was permitted by the federal government (Table 1 & [16]). Therefore, we had to contend with a pseudo-control as follows: At times, the state agency opted not to kill wolves or opted to offer farmers non-lethal deterrents, and the state advised the complainant on protection of livestock. The latter intervention involved communications and possible deployment of non-lethal deterrents (see below) with unknown characteristics or consistency. We also considered potential measurement errors–that may have been systematic, not random errors–associated with unreported wolf-killing and unreported depredations, both of which occur in neighboring Wisconsin [2, 14], and are believed to occur in Michigan as well [17, 18].

Table 1. Periods for wolf-killing policy signals in WI and MI, derived from Refsnider [16], ESA sec. 4 10(a)(1)(A) and Humane Society of the U.S. et al. v. Jewell (U.S. District Court, D.C., 5 1:13-cv-00186-BAH Document 52, 2014).

Period start (mm/dd/yyyy)	Period end (mm/dd/yyyy)	Federal status	Culling**
4/15/1994	3/31/2003	Listed as endangered	not allowed
4/1/2003	1/30/2005	Down-listed to threatened	allowed
1/31/2005	3/31/2005	Relisted	not allowed
4/1/2005	9/13/2005	Sub-permit for culling issued	allowed
9/14/2005	4/23/2006	Sub-permit rescinded	not allowed
4/24/2006*	7/31/2006	Sub-permit for culling issued	allowed
8/1/2006	3/11/2007	Sub-permit rescinded	not allowed
3/12/2007	9/28/2008	Delisted	allowed
9/29/2008	5/3/2009	Relisted	not allowed
5/4/2009	6/30/2009	Delisted	allowed
7/1/2009	26/1/2012	Relisted	not allowed
1/27/2012	4/14/2012	Delisted	allowed

*States identical except sub-permit issuance on 6 May 2006 to Michigan instead of issuance on 24 April 2006 to Wisconsin [16].

**Killing a wolf that posed a threat to human safety was always allowed under ESA sec. 11(a)(3).

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Selection bias–or the tendency to apply different interventions to different subjects or locations based on some anticipated outcome–can powerfully affect the results of experimental tests [5]. In short, we controlled for spatial variation by comparing an intervention site to itself, but we could not control for the intervenors' subjective decisions. In the Discussion, we identify and discuss potential sources of bias in the dataset provided to us.

Because of the caveats above relating to the strength of inference we might draw from the uncontrolled 'experiment' conducted by the State of Michigan, we regard our conclusions as preliminary in the same way that other recent published studies should be considered, pending gold-standard experiments [4, 9, 12]. These studies offer new inferences and testable hypotheses about the effect of interventions, rather than conclusions about the functional effectiveness of the interventions *per se*.

Materials and methods

Data sources

The State of Michigan continuously monitored complaints about wolves and annually monitored the wolves themselves, across the Upper Peninsula (42,610 km^2). We used the federal government's published reports for Michigan's minimum, late-winter wolf population (https://www.fws.gov/midwest/wolf/aboutwolves/mi_wi_nos.htm), supplemented by Michigan data provided to the Little River Band of Ottawa Indians after their request through a federal Consent Decree. Michigan estimated wolf numbers by snow-track surveys, summer howling, and aerial telemetry of VHF radio-collared wolves primarily [19]. The exception was wolf-year 2012 when Michigan did not census its wolf population, so we interpolated the midpoint of the 2011 and 2013 estimates (Fig 1). Our study spanned wolf-years 1998–2015 (calendar-years 1998–2014); a wolf-year t was 15 April of year t-1 to 14 April of year t.

Michigan provided Wolf Activity Reports with 379 entries. The U.S. Department of Agriculture Wildlife Services (USDA) investigated many of these incidents since 1990 under state contract [20]. Hereafter, we refer to Michigan when referring to government responses to wolf-related complaints, whether by state or USDA field personnel. We discarded 149 entries that consisted of different categories of wolf encounters: observations, perceived threats to

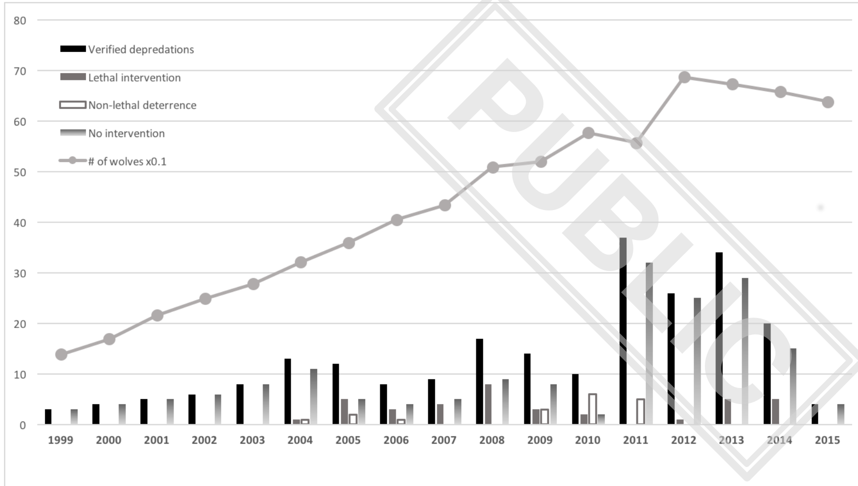

Fig 1. Annual Michigan wolf abundance, verified depredations and interventions. Michigan's annual wolf abundance (divided by 10 to fit the same y-axis as other variables) and two treatments after verified depredations. The x-axis shows wolf-years, which span 15 April of year t-1 to 14 April of year t. Overall n = 230 depredations.

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humans or domestic animals, or wolf interactions with hounds engaged in training or hunting, but which lacked verified depredations on a private property. Discarding perceived threats to humans should prevent the introduction of some biases, because the Wolf Activity Report entries suggested that one complainant's 'threat' was another's 'encounter' that did not result in official complaint, investigation or intervention. Considering the potential biasing effects of perceived threats that did not lead to a complaint (false negatives), perceived threats that were simply observations (false positives), and a complete lack of any such reports before 2002, we felt more secure setting aside all entries lacking depredations and ensuing verification. Also, wolf interactions with hounds occur under very different circumstances than depredations in our region [21–24]. In sum, we retained 230 complaints for screening as described below.

We screened complaints for verification and independence between depredations. During the study period, Michigan verified 499 livestock or farm animals injured or killed by wolves in 230 complaints. Depredations were classified as independent if they occurred on a different date.

Michigan responded in several ways to predation: communication only, provision of non-lethal deterrents, or lethal intervention. Lethal intervention consisted of live-trapping on or near the complainant's property for several days to weeks after a depredation, and if successful, the state shot one or more wolves caught alive in leg-hold traps (n = 98 wolves killed overall, with lethal interventions following depredations in 37 occasions, and resulting in the deaths of 56 wolves in 32 interventions and 0 wolves killed in 5 occasions); in a few cases landowners shot wolves after receiving state permits. We omitted 32 cases in which wolves were killed but were not involved in depredations; only two of which occurred in the same townships (geopolitical mapping area of 36 miles^2 or 92.16 km^2) as lethal intervention during our study. We did not include the public hunting season at the end of 2013 because those removals were not targeted at known complaint sites [25]. Non-lethal deterrence was used primarily when no losses occurred in the Wolf Activity Reports, so most such interventions were excluded by our screening criteria above.

We refer to any intervention that did not lead to wolves dying as non-lethal, which implies only that no wolves were killed, but related actions may have entailed a range of communications with the complainant and other responses, including the provision of non-lethal deterrents in some cases. All interventions included communications with complainants but we had no data to determine if such communications differed between lethal intervention and non-lethal. Non-lethal deterrents included one or more of the following: cracker shells, hazing kits, live-traps, lights, or fencing with various materials, including fladry (a loose flagging hung at regular intervals on fence-lines [26]). We also classified live-trapping (i.e., attempted lethal interventions) that resulted in no wolves killed (n = 5) as 'non-lethal'. Differences in non-lethal methods implemented at different sites could be attributed to costs, judgments by state agents about effectiveness in a given situation, willingness of livestock owners to deploy certain techniques, or other undocumented factors. Because of the small sample of occasions when non-lethal deterrents were deployed after depredations (n = 18), we pooled all interventions that did not lead to wolf-killing as non-lethal, due to insufficient information on whether the deterrents were actually implemented by the farmer.

A true control would have enacted all the same procedures and time spent on the complainant's property without killing wolves, or installing any non-lethal infrastructure. Therefore, we refer to our non-lethal intervention classification as a pseudo-control because it may have included different communications or a judgment by a state agent that lethal intervention was not likely to succeed. However, given that the federal permit for the state to use lethal control was issued and rescinded several times without regard to events on the ground (Table 1), we infer that the two treatments we analyzed were largely selected because of the broader governmental timelines rather than the events at a particular property. Independent decisions about the availability of lethal intervention would reduce the risk of treatment bias [2]. Regardless, this study represents a silver-standard experiment with possible treatment biases that must be considered preliminary and examined carefully (see Discussion).

With the preceding criteria, our primary sample of 230 depredations (or depredation events, by which we mean a verified, independent wolf depredation incident in the Wolf Activity Report) consisted of 32 depredations followed by lethal intervention, and 198 followed by non-lethal intervention.

Analyses

We used geopolitical sections (regular units of 1 mile^2 or 2.56 km^2) as the smallest mapping units, following [27]. Sections can be read from commercially available road atlases. Sometimes more precise locations were also provided, but inspection revealed that many of these were simply the latitude and longitude of the center of the section. Virtually every livestock pasture lay within the borders of a single section. All livestock pastures were on private property of much less than 1 section in area (average farm size was 0.3 miles^2 or 0.68 km^2 in the Upper Peninsula [28]). The state did not record ownership of pastures or the tenure status of complainants. All depredation events are presented in **S1 Data File** with certain personal details, property information, and precise locations redacted for privacy.

We determined the sequence of depredation events by reference to the date of the complaint on the Wolf Activity Reports. We calculated the delay to recurrence as the interval in days to the next event in the same vicinity (2.56 km^2 section or larger geographic unit, see below). If there were no subsequent events in the vicinity that calendar year, we censored that observation of delay to recurrence at 31 December of the same year. Virtually all depredations occurred in the warmer months [20], with most events occurring in the period March-October (90%) and only 3% occurring in November or December, echoing results from Edge et al. [18]. Livestock in the Upper Peninsula are kept within enclosed pastures year-round, usually in small farms, and thus equally available to wolves throughout the year [18, 29]. Therefore, our decision to measure and censor the delay to recurrence within the calendar year provided at least 60 days to detect an effect in 97% of events (recurrence at section scales occurred within a median of 13 days if it occurred the same year). Had we extended the time horizon as in [4], we saw a risk of conflating the recurrence of depredation events by later wolves with the treatment applied to prior wolves.

We also examined if depredations recurred at two larger spatial scales. At the intermediate scale of townships (36 miles^2 or 92.16 km^2), the area used for measuring recurrence approximated half the core area of an average wolf pack territory [30]. At our largest spatial scale, the neighborhood of townships (320 miles^2 or 829.44 km^2) was equivalent to 9 contiguous townships centered on a depredation event and >4 times the average core area of a wolf pack territory [30]. For analyses of risk of recurrence at the township and neighborhood scales, we replaced the fixed geopolitical unit with a square buffer of the same area centered on each depredation event (Fig 2). We detected no difference in the sequence of depredation events for particular areas when using a circular buffer, possibly due to the coordinates for depredation incidents obtained from the Wolf Activity Reports frequently placing the incident in the center of a section, which both buffer shapes contained. The square buffer was preferred based on its consistency with the underlying Public Land Survey System (USGS, https://nationalmap.gov/small_scale/a_plss.html) layer containing the spatial subdivisions we based our three spatial

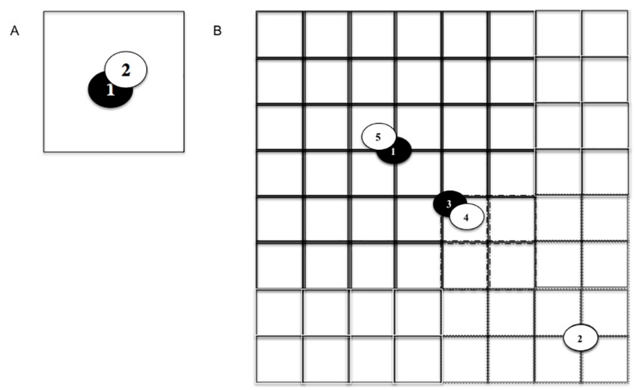

Fig 2. Measuring recurrence between depredation events at multiple spatial scales. Each small rectangle is a section (1 mile^2). Each oval is a single event of verified depredation. A 1 indicates the first of such events in its vicinity and year, and higher numbers are subsequent events in chronological order of occurrence in the same year. The intervention is shown with colored ovals: lethal (black), non-lethal intervention (open); and events within the same section are depicted as overlapping each other partially (1 and 2 in A; 1 and 5 or 3 and 4 in B). **A**: Smallest scale of analysis where the vicinity is limited to the section. Datum 1 stratum 1 measures the number of days between events 1 and 2 with lethal intervention. Because there is no event 3 within the vicinity, datum 1 stratum 2 measures the number of days between event 2 and the end of the calendar year but switches to non-lethal intervention (open oval). **B**: Medium-scale of analysis where rectangles are sections in a township (36 miles^2 centered on event 1). Solid black grid lines indicate buffer around event 1; dotted gray lines indicate buffer around event 2; black dot-dashed lines indicate overlap between buffers. Because event 1 and event 2 are not in the same township-sized buffer, they generate datum 1 and datum 2 with lethal intervention and non-lethal intervention, respectively. Datum 1 stratum 1 measures the number of days between event 1 and event 3. Although event 3 is also within the buffer of event 2 (within black dot-dashed lines), it was assigned to event 1 because it was nearest by Euclidean distance. We did not measure the number of days between events 2 and 3 because event 3 was already used to create datum 1 stratum 1; in this way, we avoided double-counting events. Next, events 3 and 4 are collapsed (treated as a single event) because they occurred in the same section *sequentially*. Because event 3 was followed by lethal intervention (black oval), the resulting single collapsed event was classified as lethal intervention. We then measure datum 1 stratum 2 as the number of days between event 4 and 5, remembering that the collapsed event is classified as lethal even though 4 is followed by non-lethal intervention (any collapsed set of events with a lethal intervention event among them is assigned to the lethal intervention set). Finally, datum 1 stratum 3 is measured by the number of days between event 5 and the end of the calendar year and assigned to non-lethal intervention. If event 2 had zero other events in its township area (not shown), then datum 2 stratum 1 would be measured to the end of the calendar year. A similar process was followed for the largest spatial scale of neighborhood of townships (320 miles^2).

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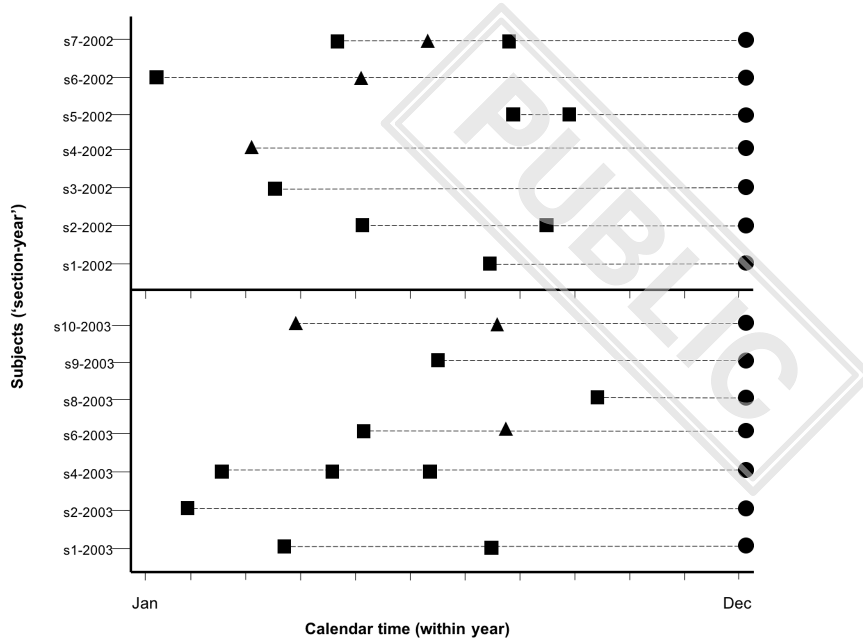

Fig 3. Transforming depredation records to a survival analysis format. We present lethal interventions (triangles) and non-lethal interventions (squares) connected by a dashed line that measures the delay to recurrence or censorship (circles). We illustrate using data from two subsequent years. Subjects are identified as combinations of vicinity (section, township or neighborhood) and year (i.e.: section s1-2002) on the y-axis. The first figure for each subject represents when the first depredation event in that year occurred, which is the date follow-up started for that 'section-year'. Each subject then follows a chronology of subsequent depredation events through the year, treated with either intervention. Stratum 1 considers the initial intervention implemented and the delay to recurrence to the next depredation event, or censoring if no other events occurred (i.e.: first figure to second figure in dashed line for each subject). Stratum 2 considers the next sequence of depredation events (i.e.: delay from second figure to third figure). Due to our construction of subjects, a particular section (sections 1, 2, 4 and 6, for example) can appear in multiple years, represented with a different 'section-year' combination (for example, s1-2002 and s1-2003).

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scales on. We measured delay to recurrence in that buffer, repeating the process for subsequent depredation events using each event's buffer (i.e., a moving window). The process of assigning depredation events at scales larger than the section (Fig 2) was designed to avoid pseudo-replication (once the effect of a pair of events was measured at a lower scale, that estimate of delay to recurrence was never used again at larger scales).

The use of three spatial scales allowed us to detect depredation recurrence beyond the original sites (spill-over effects) following interventions. Our process for collapsing depredation events (Fig 2) produced a conservative assessment of spill-over effects because we eliminated pseudo-replication of estimates of risk of recurrence across scales. The disadvantage of our approach was declining sample sizes that reduced the power of the tests at larger scales and thereby potentially increased Type II error.

Statistical tests

We measured delay to recurrence in days between each pair of successive depredation events as in Figs 2 and 3, and produced survival functions for each treatment following Hosmer, Lemenshow & May [31]. A survival function describes the probability of observing a time interval between two depredation events, T, greater than some stated value t, $S(t) = P(T>t)$, where t is days. Thus, survival functions provide, for every time t, the probability of 'surviving' (in this case, not experiencing a depredation event) up to that time, and describe these probability distributions (survival distributions). Survival analysis comprises a set of statistical methods used to quantify and test survival function differences between treatment groups of subjects [32].

At the smallest spatial scale, we defined our subjects as the sections in which depredations occurred. Thus, sections are analogous in biomedical research to the patient receiving treatments. In this case, the section receives lethal or non-lethal treatment of wolves. Note that this differs from prior research that defined wolf pack territories as the subjects [4].

Subjects enter the analysis after the initial depredation event, and remain in the analysis until December 31st of that year; hence, our subjects arise from a particular vicinity (i.e., section, township or neighborhood) in a particular calendar-year (1998 to 2014) (Fig 3). Depredation events, along with their respective treatments and measures of recurrence were organized into strata based on their order of occurrence for each subject (Fig 2). Each year a new set of strata was created, starting with stratum 1 again. The end of each calendar year represented a 'reset' point after which we assumed independence of subjects because both wolves and livestock are mostly removed from each other's reach until the next grazing period. Based on this classification of subjects and strata, we clustered our analysis on a unique identifier reflecting a particular vicinity-year combination, e.g., ID_TRS_Yr [33]. This approach accounts for potential spatial and temporal auto-correlation among strata within subjects, e.g., all depredation events for the same subject experienced during a particular year are assumed correlated. It also avoids pseudo-replication of observed depredation events from the same subject as if they were independent of other depredation events in that same year, e.g., ID_TRS_2000's stratum 1 and stratum 2 observations are correctly identified as belonging to the same subject, rather than belonging to two different subjects (pseudo-replication). In the Discussion, we examine potential pseudo-replication concerns in our dataset and in prior approaches.

We employed general and stratified log-rank tests (Chi-squared statistic) to compare the survival distributions for delay to recurrence in both treatments. We then used a conditional Cox recurrent event, gap time model [31] to compare the associations between treatments and risk of recurrence. The Cox model allowed us to estimate hazard ratios (HR) for relative risk of recurrence between treatments by characterizing how the hazard function (H) changed as a function of survival time and subject covariates; $S(t) = e^{-H(t,x,ß)}$, where t is study time (the period of observation or follow-up period after inclusion in study until end of the calendar year), x is a covariate we describe below, and $ß$ is the parameter estimate of x.

The *stratified* conditional Cox model accounts for risk of recurrence for the ith depredation event being influenced by the occurrence of a previous (i-1)th depredation event and the treatment following it, so that each subject is included in the risk set (the number of subjects experiencing a depredation event) for the ith depredation event only if it experienced the (i-1)th depredation event. For example, in our section-scale analysis, 31 subjects experienced a first recurrent depredation event, whereas 120 did not experience any recurrence (Stratum 1, Tables B & C in S1 File).

The stratified Cox model considers only those subjects experiencing that first recurrent depredation event in the second stratum (Stratum 2, n = 31; Table A in S1 File), repeating the process for subsequent strata until end of the calendar year. The stratified Cox model allowed us to estimate general treatment effects while accounting for event order and the treatment applied to the previous event.

We ran univariate and multivariate conditional Cox models at each spatial scale. Univariate models included only our response variable (delay to recurrence) comparing our two treatments, whereas multivariate models incorporated calendar year. Including calendar year was essential because the gray wolf was down-listed to threatened in Michigan on April 1, 2003, and subsequently went through 12 or more reclassifications and permit issuances that precluded or allowed wolf-killing by the state ([34], and Table 1]) as the protection afforded wolves was reduced or increased.

Given that treatment effects could change over time as wolves, livestock, people, and ecosystems might change with environmental conditions, we also ran multivariate models incorporating a time-varying covariate (tvc) for treatments [31]. Our tvc consists of an interaction of treatment with study time. The use of a tvc is strongly recommended for evaluating and handling non-proportional hazards (PH), given PH is an underlying assumption of survival modelling [31]. A non-proportional hazard occurs when the treatment effect changes over time (instead of remaining constant) relative to the pseudo-control, so that the hazard ratio for the treatment changes over time. Hence, if the parameter estimate for the tvc were found to be significant, the conditional Cox model with tvc would be more robust and reliable than without the tvc because it corrected for non-proportional hazards in our treatments. When the tvc is not significant, its inclusion in the model is not warranted.

Authorities on stratified Cox models also express concerns about strong inference depending on the risk set per stratum [31, 35]. The latter authors did not settle on a particular number observations per treatment per stratum; however, the Cox models depend on a measure of variability within-strata to detect deviations from chance differences between treatments, therefore we excluded strata with <10 depredation events or which lacked events for both treatments. This conservative step left us with 3 strata at the section scale, 1 stratum at the township scale, and 2 strata at the neighborhood scales (S1 File). Thus, our final sample at the section scale consisted of 151 subjects (independent section-years) with 199 depredation events, including 56 recurrent depredation events; the final sample at the township scale consisted of 125 subjects with 125 depredation events, including 24 recurrent depredation events; and the final sample at the neighborhood scale consists of 106 subjects with 125 depredation events, including 25 recurrent depredation events (S1 File).

We assessed the robustness of models to within-subject correlation by running a variant of a random-effects approach called frailty models ([35]; S2 File). If high-risk and low-risk farms exist due to factors extrinsic to treatments, years, or the tvc, then subject identity should inform gap time models [17, 36]. Frailty models assess the goodness of fit of the treatment variable by including random effects of subject identity [35], which is considered useful when recurrence time might be influenced by unmeasured factors [31, 37].

We also built models with subsets of the data to evaluate potential confounding effects and robustness of the primary models described above. We built a model with data 'post-2003', after lethal management was episodically permitted, and by reclassifying lethal management with zero wolves killed as 'lethal' because the infrastructure and attendant human influences would be the same whenever traps were laid regardless if wolves were live-trapped and killed. We refer to the latter condition as 'traps placed'. We present alternative models in supporting information (S2–S4 Files).

Finally, we used Spearman rank correlations (r_s) to correlate delay to recurrence with number of wolves killed for lethal treatments only and for 'traps placed'. We conducted all analyses in Stata 14 (StataCorp, College Station, TX, 2015; protocol DOI: 10.17504/protocols.io.j2rcqd6).

Results

Between 1998 and May 2014 there were 199 depredations in Michigan with as many management interventions. Of the 199, 31 resulted in lethal intervention (16%) and 168 resulted in non-lethal intervention (84%) (Fig 1).

Table 2. General and stratified log-rank (χ^2) tests examining difference between treatments' (lethal and non-lethal) survival distributions (measuring risk of recurrence) after wolf depredations, for all spatial scales.

	Spatial scale of analysis		
	Section	Township	Neighborhood
SUBJECTS AND 'FAILURES'			
TOTAL DEPREDATION EVENTS	199	125	125
Failures (recurrent events)	56	24	25
SURVIVAL FUNCTIONS			
Log rank test (χ2)	0.27	1.44	0.08
p-val	0.603	0.23	0.772
Stratified Log-rank test (χ2)	0.48	-	0.28
p-val	0.488	-	0.593

https://doi.org/10.1371/journal.pone.0189729.t002

Section scale

Log rank tests could not distinguish the survival functions between treatments (df = 1, general survival functions test: $\chi^2 = 0.27$, P = 0.604; stratified [by order of depredation events for subjects] test: $\chi^2 = 0.48$, P = 0.488; Table 2). All univariate (treatment only) and multivariate (treatment and calendar-year) Cox models suggest that lethal intervention was associated with a non-significant reduction in risk of recurrence when compared to non-lethal intervention (Table 3). The section-scale models including a time-varying covariate (tvc) were not significant, so the PH assumption was not violated (tvc P>0.05). The multivariate model including treatment and year suggests lethal intervention only weakly reduced risk of recurrence (slowing recurrence) by 27%, but that was not a statistically significant difference (HR = 0.73, P = 0.326; Table 3). This model also revealed an increasing risk of recurrence (hastening recurrence) by 9% each calendar-year (HR = 1.09, P = 0.022). Lethal intervention was not significantly different from non-lethal intervention in our frailty model (HR = 0.48, P = 0.158; Table A in S2 File), with the model suggesting significant frailty (omitted or unobserved

Table 3. Main results of Cox models measuring risk of recurrence between treatments (lethal and non-lethal) implemented after wolf depredations, for all spatial scales.

	Spatial scale of analysis					
	Section		Township		Neighborhood	
PROPORTIONAL HAZARD MODELS	*Interv*	*Interv & year*	*Interv*	*Interv & year*	*Interv*	*Interv & year*
Standard cox (stratified)						
Intervention HR (SD)	0.77 (0.22)	0.73 (0.23)	0.48 (0.308)	0.46 (0.29)	0.80 (0.340)	0.72 (0.34)
p-val	0.36	0.326	0.255	0.224	0.644	0.486
year HR (SD)	-	1.09 (0.04)*	-	1.05 (0.05)	-	1.14 (0.07)*
p-val	-	0.022	-	0.28	-	0.024
Standard cox with tvc (stratified)						
Intervention HR (SD)	0.48 (0.21)*	0.46 (0.21)	1.87 (1.47)	1.78 (1.38)	0.84 (0.62)	0.80 (0.63)
p-val	0.099	0.091	0.425	0.458	0.818	0.778
tvc(Intervention) HR (SD)	1.01 (0.01)*	1.01 (0.01)	0.97 (0.01)**	0.97 (0.01)**	0.99 (0.01)	1.00 (0.01)
p-val	0.057	0.068	0.001	0.001	0.928	0.852
year HR (SD)	-	1.09 (0.04)*	-	1.05 (0.05)	-	1.14 (0.07)*
p-val	-	0.023	-	0.281	-	0.023

Significance

* if p-val < .05

** if < .01.

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Table 4. Spearman correlation between delay to recurrence and number of wolves killed after depredation events followed by lethal intervention (wolves killed ≥ 0), for all spatial scales.

	Section	Township	Neighborhood
Spearman's rho	0.107	0.212	0.295
p-val	0.5591	0.2994	0.1354

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covariates) remaining in the model (P = 0.006). For those depredation events followed by lethal intervention, we found no correlation between delay to recurrence and the number of wolves killed (Spearman's rho = 0.107, P = 0.559, Table 4; 'traps placed': Spearman's rho = 0.076, P = 0.657; Table C in S3 File).

Township scale

Our dataset consisted of 125 depredations, 26 followed by lethal intervention (21%) and 99 followed by non-lethal intervention (79%). Log rank tests could not distinguish the survival functions between treatments (df = 1, general test: $\chi^2 = 1.44$, $P = 0.23$; Table 2). Likewise, all Cox models revealed no significant differences between treatments (Table 3). The township-scale models including a tvc were significant, suggesting the PH assumption was violated (tvc P<0.05). Hence, we focus our analysis on the model including the tvc. Lethal intervention increased risk (hastening recurrence) by 22%, but this was not statistically significant (treatment HR = 1.78, P = 0.458). However, our tvc, which accounts for non-proportional hazards, hints at a minimal (3%) reduction in risk over follow-up time (tvc HR = 0.97, P = 0.001). Calendar-year was not significant (HR = 1.05, P = 0.281). Differences between treatments were not significant in our frailty model (HR = 0.45, P = 0.242; Table A in S2 File). For those events followed by lethal intervention, we found no correlation between delay to recurrence and the number of wolves killed (Spearman's rho = 0.212, P = 0.299, Table 4; 'traps placed': Spearman's rho = 0.233, P = 0.224; Table C in S3 File).

Neighborhood scale

Our dataset consisted of 125 depredations, 26 followed by lethal intervention (21%) and 99 followed by non-lethal intervention (79%). Again, log rank tests could not distinguish survival functions between treatments (general test: $\chi^2 = 0.08$, $P = 0.772$; stratified test: $\chi^2 = 0.28$, $P = 0.594$). Similarly, all Cox models revealed no differences between treatments (Table 2). The neighborhood-scale models including a tvc were not significant, so the PH assumption was not violated (tvc P>0.05). Lethal intervention only weakly reduced the risk of recurrence (slowing recurrence) by 28% but this difference was not significant (treatment HR = 0.72, P = 0.486; Table 3). We found a statistically significant increase in risk of recurrence (hastening recurrence) of 14% every calendar-year (HR = 1.14, P = 0.024). The frailty model showed no significant differences between treatments (HR = 0.80, P = 0.67; Table A in S2 File).

For those events followed by lethal intervention, we found no evidence of a correlation between time to recurrence and the number of wolves killed (Spearman's rho = 0.295, P = 0.135, Table 4; 'traps placed': Spearman's rho = 0.161, P = 0.395; Table C in S3 File).

For all spatial scales, all effects of treatment remained consistent for the 'traps placed' condition, when limiting the data to post-2003 depredation events, 'skip-a-year' dataset and when removing a special case (S2–S4 Files).

Discussion

We retrospectively evaluated whether lethal interventions by the State of Michigan in response to wolf predation on domestic animals (depredations) between 1998–2014 resulted in lower risk of recurrence of depredations than if no wolves were killed. We found the delay to recurrence of depredations was unrelated to the number of wolves killed at all spatial scales. We found lethal management did not significantly shorten or lengthen the interval to the next depredation relative to non-lethal interventions. A small, statistically insignificant reduction in the risk of depredation at the section level was offset by a similar and also statistically insignificant increase in the risk of depredation at the township scale, which is about half the size of a wolf pack territory, and then a similar decrease in risk at the scale of neighborhoods of townships, which are four times larger than the average wolf pack territory [30]. None of these differences were statistically significant using a battery of tests.

Our methods or alternative models accounted for potential violations of the proportional hazards assumption, unlike a prior study of wolves in the Northern Rocky Mountains (see below); accounted for within-subject correlation; were unaffected when we restricted analysis to the period after 2003 when lethal interventions first became legal; and accounted for a change in definition of lethal methods to include the installation of lethal methods that did not kill any wolves (S3 File). There is evidence for the effect of lethal intervention changing slightly over the course of a single calendar year at the township scale, through a minimal reduction in risk over follow-up time. We also detected variation between individual farms in their time to recurrence of depredations. Given the apparent, net ineffectiveness of lethal intervention and the uncertainty about potential biases in a retrospective analysis of sparsely documented government interventions, we recommend ethical, gold-standard, random-assignment experiments be used before further lethal management is authorized to prevent depredations.

Overall, our analysis suggests that any potential beneficial effects of lethal interventions locally would be offset by detrimental effects for neighboring farms in the same township. If the small, local improvements were considered biologically, ethically, or economically important to one farm, then one would also have to admit the associated costs to neighboring farms and the biological, ethical and economic importance to that farm. Therefore, given the evidence available, we cannot conclude that lethal management had the desired effect of preventing future livestock losses.

Over the 17 years of our study, the risk of depredation increased by 9 and 14% per year at the section and neighborhood (smallest and largest) scales, respectively, in our main dataset. However, this effect of year is insignificant in our post-2003 dataset (S4 File). In addition to changes in wolf densities locally that may have occurred, there may also have been changes in proportion of pasture, prey density, land cover, farm size, road density, among other variables that predict depredations at local scales [17, 38]. Also, prior work indicated smaller packs were more often implicated in livestock depredations than larger packs [23]. Therefore, the notion that higher densities of wolves locally will result in more depredations is not well supported, as opposed to the idea that a recolonizing population encounters more livestock as a result of recolonizing more and more of their historic range over time.

We present our results guardedly rather than as a definitive conclusion about effectiveness because of insurmountable uncertainties about the government data. Retrospective analyses to evaluate the effectiveness of interventions to prevent predation on livestock are fraught with uncertainty because of various biases or challenges presented by field conditions [2]. For example, treatments were not assigned randomly and changing conditions over time locally were not documented. The unintentional error may have been random but we are unable to rule out systematic error (bias), whether intentional or unintentional. The government dataset we analyzed had undocumented variability in data collection and intervention, including possible systematic selection bias affecting which areas received which interventions.

Selection (or enrollment) bias would arise if subjects entered the study under varying conditions that affected outcomes. All sections containing farms (subjects) entered our study because of a verified depredation, but subjects entered at different times and some farm owners might have responded to depredations in undocumented ways including poaching wolves. Likewise, attrition bias would arise if subjects left the study for reasons that were not random with respect to their outcomes. This would occur systematically if a subset of the interventions led farmers not to complain in the future despite facing depredations, or to take matters into their own hands, as above. Compensation was offered throughout the study as well as state-financed non-lethal deterrence when lethal intervention was unavailable, so attrition by withholding complaints seems unlikely to have been frequent or widespread. However, we would guess that non-intervention might be construed as unhelpful by complainants, leading some of them to intervene independently. We consider unreported wolf-killings to be a more pronounced confounding variable after 2003, when state lethal management was allowed (Table 1), substantiated by a recent inference that allowing state killing of wolves seems to have potentially increased poaching of Michigan and Wisconsin wolves [34]. By definition, poaching can only confound tests of non-lethal deterrence because poaching following lethal intervention would only increase the number of wolves killed (undetectably in our context), but not change the nature of that lethal intervention. We do not see how poaching could confound the apparent reversal of effects of lethal control across our three geographic scales of analysis.

Furthermore, treatment bias would arise if methods of intervention were not standardized. Treatment bias certainly arose among non-lethal deterrents because different complainants received different types of non-lethal methods and we do not know if they maintained or installed the methods appropriately or identically. Non-lethal deterrents were presumably negotiated with complainants and therefore most prone to treatment bias that would confound our results. However, only 8% of our eventual sample received non-lethal deterrents. Moreover, we have no data on other deterrents or precautions unilaterally implemented by complainants. Lethal interventions were more uniform in method [20] but we did not receive precise, detailed information on implementation (number of trap-nights, exact locations, etc.). Moreover, if lethal interventions were spatially segregated from other types of interventions, then selection bias might have applied systematically because farms perceived to be higher-risk might have received lethal interventions preferentially and also be expected to have recurrent depredations. This might have resulted in significant, between-subject variability. Such a bias would not explain the spill-over effect we detected. Intermittent authority for lethal intervention led to the same spatial units receiving all types of intervention (**S1 Data File**). Given that authority for the state to kill wolves after verified depredations was granted or withheld by federal decisions unrelated to area attributes or recent depredation complaints and in several years of the study even high-risk areas received no interventions [16], it seems unlikely that lethal control authority for Michigan coincided with risky years. Therefore, any treatment bias (intervening lethally at sites that were inherently more likely to have recurrence of depredation) would have to occur at the spatiotemporal scale of individual farms within years. We addressed within-subject variability using a frailty model (S2 File), which revealed the presence of confounding effects at the section level, but the treatment effect remained statistically insignificant.

Finally, wolf abundance was unlikely to confound our tests because the number of wolves within our spatial units was unlikely to change substantially from one incident to the next within a small area within one year.

In sum, we find ample reason to expect confounding variables would weaken inference from a retrospective, quasi-experimental test of interventions to prevent livestock loss. Our attempts to detect and screen for biases were necessarily imperfect because we could not assign treatments randomly nor could we retrospectively assess if interventions were assigned haphazardly or subjectively. Our analyses controlled for variation in risk due to time and inter-farm differences using tvc and frailty models (S2 File), but could not ultimately control for transient changes in risk associated with wolves, people, or other wildlife. Moreover, we were not able to account for illegal wolf-killing that might have added to treatment bias affecting non-lethal interventions.

Nevertheless, there is value in the scientific examination of on-the-ground programs of predator management as they are actually carried out by the organizations that discharge them. Avoidance of selection, treatment or measurement biases would require enforcement of strict protocols that are rare worldwide [2, 39–41]. In addition to understanding how the strongest inference arises from gold-standard experiments without bias, wildlife managers have a responsibility to continually evaluate their particular actions and policies to ascertain if they are effective at accomplishing the goals set by the broadest society, and to remedy or terminate them if they are found to be ineffective, as evidence-based policy-making demands.

An example of a gold-standard design that might achieve strong inference would be random-assignment of treatment to different, large areas (e.g., 324 km^2) with uniform treatments, in which measurement is unbiased by blinding or independent, third party monitors, and data analysis is conducted by independent, third-party analysts without financial conflicts of interest involving the government or livestock industry. However, such an experiment would have to address the ethical implications for both animals and people of removing wild animals, possibly exposing more livestock to spill-over effects, and the broad public interest in preserving both wildlife and livelihoods. A step in that direction, albeit imperfect, may be to temporarily relocate predators to captivity until the analysis period ended in each area.

If our results are supported by a gold-standard experiment, we propose a hypothesis for two long-standing phenomena about human perceptions of conflicts with predators and the perceived effectiveness of interventions. We observe that killing predators is widely perceived to be effective (e.g., in our region: [42, 43], yet afterwards real and perceived risks appear to increase [44]. The spill-over effect may be responsible. Our hypothesis builds on the idea first articulated by Haber [45] that killing wolves can trigger pack disruption which might lead to more livestock predation than done by intact packs. If our inference about spill-over effects is confirmed, then we hypothesize that the perceived effectiveness of lethal methods stems from a few livestock owners who report preventive benefits, while neighboring livestock owners report increasing losses because of the spill-over effect from the former farms. The adverse effects of killing wolves as a response to depredations might thereby be obscured by anecdotal accounts and misperceptions.

Our results appear to contradict those of the [4] in the Northern Rocky Mountains (NRM) for the period 1989–2012. Although [4] conducted similar survival analyses, they found lethal methods significantly reduced the risk of recurrence, and that killing an entire wolf pack was more effective than the killing of a subset of members of a pack. They reported only a marginal difference between partial pack removal and no removal if wolves were killed within the first 7 days following a depredation event and no difference if 14 days elapsed. Most lethal interventions in Michigan were probably partial pack removals (median wolves killed = 1, **S1 Data File**) so our results are consistent. However, other differences in results between their study and ours could be due to different sites and methods.

The analysis in [4] included more varied methods of lethal intervention and the landscapes differ (theirs being mountainous and wider while Michigan's is flatter and surrounded by water on three sides, with attendant differences in vegetation, lake effects, human population density, wolf migration, livestock husbandry practices, etc.). In addition, the survival analyses employed by [4] differed from ours in ways that we could not resolve despite several email exchanges with the lead author and the analyst co-author.

First, [4] did not account for treatment effects beyond a single spatial scale (see Box 1). Their analysis was restricted to the affected wolf pack territory, despite their own reports that killing wolves had at times scattered surviving pack members beyond their original territory [10, 13, 46]. This previous research would argue for an analysis that examined neighboring areas potentially affected by spill-over from scattered survivors.

Second, apparent shortcomings of the statistical modeling in [4] may have affected its results. Their measure of delay to recurrence for full pack removals spans the time from death of the last pack member to the time when a new pack attacked livestock in the same territory. This measure of delay to next depredation artificially inflates effectiveness because it incorporates a potentially long timespan before a new pack establishes, which probably includes many time-consuming events unrelated to the intervention (e.g., immigration, breeding). By contrast, our method censored observations at the end of each year, so subjects were compared on a more-equal footing after intervention. For partial removal and no removal interventions in [4], the territory was still occupied by wolves so delays probably did not include as many time-consuming demographic events (if any). Although we understand that their intent was to analyze if depredations could be delayed for longer by killing entire wolf packs, we would argue that the appropriate control for the evaluation of this intervention would be sites with suitable wolf habitat but without an established pack because of events unrelated to killing wolves, such as recolonization of vacant habitat.

Using a biomedical analogy, [4] identified the hospital bed (the pack territory) as the subject rather than the patient (the wolf pack), regardless if the wolf pack is the same or if it dies and is replaced by a new pack. Researchers continued measuring the delay to the next infection (depredation) in that bed over time, without correcting for the delay to arrival of a new patient to that bed if a previous patient dies. The delay to the next infection once a patient dies is contingent on the arrival of a new patient to that empty bed, which has little to do with the intervention implemented to the bed other than making it available for a new patient (with full pack removal). By contrast, in our study the patient (area) is the only patient, each infection receives a treatment, and delay to next infection is always measured for the same patient with a reset each year.

Third, differences with [4] could also potentially arise from different handling of the proportional hazards (PH) assumption. We evaluated the compliance of our models with the PH assumption through the inclusion of a time-varying covariate (tvc) [31]. A significant tvc affects both our treatment hazard ratios and their significance, (e.g., Table 3). We assume that [4]'s team employed other model diagnostics to evaluate their compliance with the PH assumption, but they did not report such diagnostic tests. Until the summary data are published, we cannot agree with the conclusions in [4].

Finally, some might argue that by defining our subjects as area-years and including the same area over different years we pseudo-replicated non-independent samples. In our dataset, only 16 out of 106 sections had depredation incidents in multiple years. To address that concern, we built an alternative model in which areas were omitted in succeeding years (S5 File). Results for this dataset are consistent with our main results at the section scale (S5 File).

Conclusions

Lethal interventions by the State of Michigan against wolves in the vicinities of verified livestock losses did not appear to reduce future losses. We view our findings as preliminary pending experiments with stronger inference. Our inferences could not overcome a lack of systematic information on government interventions and no effort to control for their treatments, despite a call for such shortly after the legalization of lethal removal of wolves in 2003 [47]. We detected a potential spill-over of depredations from the farm receiving lethal intervention onto neighboring farms. Given this evidence for interactions in depredations over significant areas, we must look with skepticism upon any previous or future results which analyze the functional effectiveness of lethal control but do not take these spatial relationships into account. Further, given the severe ethical issues involved in implementing harmful or lethal interventions, the lack of effectiveness of these interventions argues for their curtailing in favor of non-lethal alternatives that are effective. In the State of Michigan, there is strong scientific evidence [2] for the effectiveness of at least two non-lethal methods (fladry and livestock guarding dogs; 7–8). No peer-reviewed scientific study has ever shown lethal methods to be effective in Michigan. Indeed, our review of [4] above suggests no study in the USA has yet proven with strong inference that killing wolves is effective in preventing future livestock losses [2, 39–41]. Although it may seem obvious that killing a predator whose jaws are about to lock on a calf should protect the calf, government lethal methods are not implemented in that way. Virtually all are indirect methods such as traps placed far from the depredation site and long after a calf is killed. Therefore, rigorous scientific evaluations are a necessary prerequisite before implementing an intervention, especially given the ethical and legal obligations to balance protection of livestock and wild animals for the broad public interest. The US Endangered Species Act mandates the use of the "best scientific and commercial data available" when making conservation and management decision for listed species.

Following recommendations for ethical wildlife management [48, 49], lethal management should be discontinued, as currently the harm it causes wolves and livestock is not offset by benefits. If lethal methods are still necessary in some situations [48, 49], these should be constantly monitored and evaluated by independent third parties to measure their effectiveness or lack thereof [48].

Supporting information

S1 File. Distribution of observations and recurrent events between treatments and strata for all spatial scales.
(DOCX)

S2 File. Results from frailty mfodels for main dataset, for all spatial scales.
(DOCX)

S3 File. Results for 'traps placed' dataset.
(DOCX)

S4 File. Results for post-2003 dataset.
(DOCX)

S5 File. Results for 'skip-a-year' dataset and outlier exclusion.
(DOCX)

S1 Data File. Livestock depredation events involving gray wolves in the state of Michigan, USA (1998–2014).
(XLSX)

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