KORA Bericht Nr. 70 März 2016
ISSN 1422-5123

The recovery of wolf *Canis lupus* and lynx *Lynx lynx* in the Alps: Biological and ecological parameters and wildlife management systems

KORA

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Bearbeitung
Adaptation
Editorial

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Bezugsquelle
Source
Source

Als PDF: http://www.kora.ch

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Titelbild
Page de titre
Front cover picture

Images: © Laurent Geslin
Composition: Roland Bürki

Vorgeschlagene Zitierung/Citation proposée/Suggested citation: Breitenmoser U., Bürki R., Lanz T., Pittet M., von Arx M. & Breitenmoser-Würsten C. 2016. The recovery of wolf *Canis lupus* and lynx *Lynx lynx* in the Alps: Biological and ecological parameters and wildlife management systems. RowAlps Report Objective 1. KORA Bericht Nr. 70. KORA, Muri bei Bern, Switzerland. 276 pp.

Anzahl Seiten/Pages: 276

ISSN 1422-5123

The recovery of wolf *Canis lupus* and lynx *Lynx lynx* in the Alps: Biological and ecological parameters and wildlife management systems

RowAlps Report Objective 1

Compilation of readily available data on behalf of RowAlps Working Group 3

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Malini Pittet, Manuela von Arx
& Christine Breitenmoser-Würsten

Acknowledgments

We are grateful to many colleagues who have helped us to compile partly cryptic materials used for this report or have even provided their own unpublished data, namely:

Werner Bätzing, Piero Genovesi, Klemen Jerina, Marko Jonozovič, Harald Martens, Georg Rauer, Frank Tottewitz, Kristina Vogt, and Manfred Wölfl.

Although this report does not present any original research results, the combination of and the weight given to the various aspects considered for the assessment (Chapter 7), and the discussion and conclusions (Chapter 8) are the results of our own interpretations. To prevent a too strong influence of our own experience, we have pushed the first draft of this report through a wide review process. We are grateful to all colleagues who have commented on the first version of the report, namely:

Werner Bätzing, Luigi Boitani, Eugenio Dupré, Piero Genovesi, Marko Jonozovič, Harald Martens, Paolo Molinari, Elisabetta Raganella-Pelliccioni, Georg Rauer, Francesca Riga, Julien Transy and Manfred Wölfl.

Funding for the compilation of the reports of RowAlps Objective 1 (this report) and Objective 2 (Mondini & Hunziker 2013; Mikschl et al. 2014) was provided by the *MAVA Fondation pour la nature*.

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Executive summary

1. Lynx, wolf and brown bear are making a comeback in the Alps, forming a considerable management challenge wherever they appear. On the one hand, they are protected species in all Alpine countries; on the other hand, they cause conflicts with traditional land uses such as livestock husbandry and hunting, and fear among the local people. Another challenge is the spatial scale of (viable) large carnivore populations, which goes much beyond traditional wildlife management units and therefore calls for transboundary cooperation. The options of international collaboration in wildlife conservation across the Alps are discussed in the frame of the Platform WISO of the Alpine Convention. The project RowAlps (Recovery of Wildlife in the Alps) aims to provide scientific and technical/administrative assistance to the WISO Platform by compiling (1) relevant biological, ecological and wildlife management information, (2) reviewing human dimension aspects and factors influencing people's tolerance towards large carnivores, and (3) translating these findings into suggestions for practical management options or scenarios. This report covers the natural science part of the baseline information, with the Objective to "*review and assess, based on available scientific publications and reports, statistical materials and up-to-date experience, the present situation of wolf, lynx and their prey populations in the Alps, the expected development of the populations, and discuss challenges in wildlife management as a consequence of the return of the large carnivores*".

2. The report compiles information on wolf, lynx, their main wild and domestic prey species, and their management, habitat (e.g. forests), habitat fragmentation, the development of the human population and the human use of the Alps, which may conflict with the return and presence of the large carnivores. We performed no new analyses, but rather compared various data sets or different analyses of the same aspect and interpreted the findings and conclusions in the light of the return of lynx and wolf to the Alps. Sources of information were scientific publications and reports, and statistical material available online or provided by regional authorities and colleagues. The available information is not complete and was not gathered consistently throughout the Alps. Wherever no complete data sets were available, we use exemplary information. We tried to match the Alps (as defined by the Alpine Convention; Fig. 2.1) as close as possible and disclose where this was not possible.

3. The historic decline and eventual eradication of the large carnivores in the Alps between 1800 and the early 1900 proceeded in parallel and was related to the expanding human population and the over-exploitation of natural habitats and resources, including forests and game. Increasing numbers of sheep, goats, cattle and horses affected the forests negatively due to browsing, and out-competed the wild ungulates. The large predators were forced to kill livestock and were therefore persecuted, encouraged by governmental bounties. However, hunting alone did not lead to the eradication of the large carnivores. Only the massive intervention at the level of the landscape (forests) and the substantial reduction of wild ungulates led to the final eradication of lynx and wolf (Zimen 1978, Breitenmoser 1998a).

A radical change in forest management and the growing sensitivity of people for the protection of nature in the first half of the 20th century were the basis for the recovery of the forests (Breitenmoser 1998a). Wild ungulates started to recover and expand from remnant source populations after they were granted a certain legal protection (change of hunting legislation). Their renaissance was supported by numerous translocations and reintroductions. A swift increase in all wild ungulate populations – which is still continuing in many Alpine regions for roe deer, red deer and wild boar – was the result. The ecological recovery was facilitated by industrialisation, which drew people away from rural areas. As a consequence, the number of goats and sheep in the Alps declined drastically in the first half of the 20th century.

All these factors prepared the ground for the return of lynx and wolf to the Alps. The lynx still existed in the Carpathian Mountains, but its low colonisation capacity did not allow for a spontaneous recolonisation of the Alps. The lynx was reintroduced in the 1970s in several regions of the Alps (Breitenmoser & Breitenmoser-Würsten 2008). The return of the wolf, on the other hand, was a consequence of better protection of the remnant populations in the Apennine, in the Dinaric Range and in Eastern Europe. The first wolves arrived in the early 1990s from the Italian population and settled the south-western Alps of France and Italy.

Only the lynx reintroductions in Switzerland and Slovenia resulted in the establishment of reproducing populations. The strongest population within the Alps is located in the north-western Alps of Switzerland. Another occurrence is in the triangle Slovenia-Italy-Austria. Occurrences founded through translocations are found in eastern Switzerland and in the Austrian Kalkalpen. Scattered lynx (but without confirmed reproduction) are found in the French Alps and in the Italian Alps (Fig. 3.9).

Wolf made a remarkable come-back to the Alps. Within only two decades the species settled the French Alps and the western Italian Alps from the Italian Apennines and started to recolonise the Swiss Alps (Fig. 3.11). Wolves are also arriving from the Dinaric and Northern/Eastern European populations (Carpathian, Central European Lowland and Baltic). Austria lies in the centre of several wolf populations and could act as a cross-breeding area in the future. Thanks to non-invasive genetic monitoring, this process can be shown – given the data are processed equally between the different countries. The recolonisation of the Alps by the wolf is requiring cross-border cooperation and the regular exchange of monitoring data.

4. The SCALP (Status and Conservation of the Alpine Lynx Population) project aims for a consistent monitoring of the lynx in the Alps and developed the SCALP categories (C1-C3) to classify observations according to expressiveness and reliability. In France, a network of trained field experts is in charge of presence signs surveys throughout the year. All signs are validated by the *Office national de la chasse et de la faune sauvage* (ONCFS) (Marboutin 2013b). Signs of lynx presence in Italy are collected by a network of people, mainly game wardens and foresters who have attended special training courses. Whenever possible, they verify the signs reported by the general public. One or two persons per region are responsible for the centralisation of the data. In the Alps of north-eastern Italy, camera-traps are used to identify individual lynx (Molinari et al. 2012). Switzerland uses a stratified approach (in space, time and datasets) for monitoring. Game warden observations, chance observations, camera-trapping, radio-telemetry and genetic samples are used (von Arx & Zimmermann 2013). In Germany, experienced persons meet once a year for a country-wide assessment of large carnivore population and distribution. All lynx signs reported are inspected and documented by lynx inspectors and evaluated by experienced persons according to the SCALP categories (Kaczensky et al. 2013b). Monitoring in Austria is based on chance observations and data from opportunistic camera trapping (FaGrÖ 2009). Once a year experts meet to discuss quality and distribution of lynx sings (Kaczensky et al. 2013c). In Slovenia, monitoring is carried out by the Slovenia Forest Service using the SCALP methodology in an opportunistic manner. Genetic samples are also collected and analysed (Kos & Potočnik 2013).

Similar monitoring techniques are applied across the Alpine countries for the wolf. In France, presence signs surveys are carried out throughout the year. CMR-based estimates of abundance are derived from non-invasive sampling (genetics from excrements). Intensive snow tracking is implemented over every packs' territory. Wolf howling is implemented, when and where necessary, as a complementary field action to snow tracking, so as to get a more accurate estimate of the number of packs, and to update this number twice a year (end of summer & end of winter; Duchamp et al. 2012, Marboutin 2013a). In Italy, snow-tracking, non-invasive genetic sampling and wolf-howling are used for monitoring purposes (Boitani & Marucco 2013). In Switzerland, the monitoring is carried out opportunistically. Samples for genetic analysis, livestock and wild prey killed, sightings and pictures are collected continuously (von Arx & Manz 2013). In Germany, since 2009 the Länder have adopted country-wide monitoring standards for large carnivores using a standardised protocol including a qualification (C1-C3) of the collected data. People in charge of the monitoring in the Länder meet once a year for a country-wide assessment (Reinhardt 2013). In Austria, wolf signs are classified according to the refined German SCALP criteria. Since 2009, wolf monitoring has been based on genetic monitoring (Kaczensky & Rauer 2013). In Slovenia, monitoring included opportunistic recording of signs. In 2010 a genetic CMR method was used and systematic wolf howling introduced (Majić Skrbinšek 2013).

In 2011, the number of lynx was estimated at 130–180 individuals in the Alps (Table 4.1, Fig. 4.2). In France, the population range covered less than 1,350 km^2 in 2009, corresponding to 10–15 resident adults (Marboutin et al. 2012). In the Italian Alps 4 lynx individuals were identified in 2009. Between 2005 and 2009 signs of lynx presence and the area of presence have decreased in Italy (Fig. 4.4; Molinari et al. 2012). In 2014 two lynx were translocated from Switzerland to the eastern Italian Alps (Moinari-Jobin 2014). The population in Switzerland currently forms the largest subpopulation in the Alpine region. Between 2001 and 2008, 12 lynx were translocated from the north-western Alps and Jura Mountains to the north-eastern Swiss Alps (Zimmermann et al. 2011) to found a new population nucleus. In Liechtenstein, between 2005 and 2007 there were two records of lynx tracks and in 2008 an unconfirmed lynx sighting (Fig. 4.6; Frick 2012). There are no lynx in the German Alps (Wölfl & Wölfl 2011). In Austria, between January 2005 and December 2009, 228 records of lynx were collected of which only 14% were C1 records (Fig. 4.8; Fuxjäger et al. 2012). A female from the Swiss Alps and a female and a male from the Jura Mountains were translocated to the Kalkalpen National Park in 2011 and 2013 (Fuxjäger 2014; Nationalpark Kalkalpen 2011a, b). In Slovenia, a substantial reduction has been indicated since 2005. The current population in the Slovenian Alps was only estimated at 5–10 individuals (Kos & Potočnik 2013).

The total Alpine lynx population is still small and Endangered according to the IUCN Red List assessment. The population was stable or slightly increasing in Switzerland and stagnant in Italy, France, Austria and Slovenia. The natural fusion of the western and eastern Alpine population was concluded to be unlikely within the next decades. Persecution, low acceptance due to conflicts with hunters, habitat loss due to infrastructure development, poor management structures and accidental mortality were considered as main threats (Boitani et al 2015). Low levels of heterozygosity were found in the populations of the Swiss Alps and the Dinaric range with values below 0.5. The population in the Swiss Alps also experienced a strong genetic drift, with loss of rare alleles and changes in allele frequencies. Thus, today the population in the Swiss Alps is clearly distinct from its source population (Fig. 4.10; Breitenmoser-Würsten & Obexer-Ruff 2003). The population of the Dinaric Mountains extends north into the Slovenian Alps and is a potential source for the recolonisation of the Alps from the east. However, this population shows also a low level of genetic diversity and may suffer from an inbreeding depression (Sindičić et al. 2013). The release of a few individuals from the source population could improve the alarming genetic situation of the lynx in and around the Alps. Connecting the (sub-) populations (for example through translocations) would allow genetic exchange and thus increase genetic diversity.

In 2009/2010, the Alpine wolf population was estimated to be at least 160 wolves or 32 packs (Table 4.2, Fig. 4.12). Wolves in France are mostly found in the Alpine region (Marboutin 2013a). Census results in 2009 resulted in the identification of 13 wolf pack territories and 7 transboundary pack territories straddling France and Italy (Marboutin 2013a). By the 2009–2010 season, the population estimated through snow tracking was around 68 wolves. A wolf presence study in 2014 showed an increase of wolf presence (ONCFS 2014a). The population in the Italian Alps was estimated at 60–70 wolves in 2010-2011 (Marucco & Avanzinelli 2012), distributed across at least 12 packs, in addition to the aforementioned 7 transboundary packs shared with France. The Italian and Dinaric wolf population were separated for centuries, but in 2012 the first contact between these population was documented (Boitani & Marucco 2013). In Switzerland, a total of 60 wolves (14 females) were genetically identified from 1998-2014, but the first reproduction was only confirmed in 2012 (von Arx & Manz 2013). 24 wolves were (genetically) identified in Switzerland between October 2012 and September 2014. In Germany, between 2006 and 2011, three lone wolves were recorded in Bavaria (Bayerisches Landesamt für Umwelt 2014a). In spring 2014, two different male wolves were identified in the Bavarian Alps. In Austria, 6–8 wolves were identified in 2009 and 2010 and 2–3 in 2011 (Kaczensky & Rauer 2013). Further immigrations and the establishment of packs are expected in the near future in Austria (KOST 2012). In Slovenia (whole country), in 2010, a genetic CMR method estimated the maximum number of wolves of 43 individuals. The minimum estimate in May 2011, after the cull and before reproduction was of 32 wolves (Majić Skrbinšek 2013).

The Alpine wolf population was assessed as Endangered but with an increasing population trend. Low acceptance, habitat loss due to infrastructure development, persecution, hybridisation with dogs, poor management structures and accidental mortality were listed as the most relevant threats (Boitani et al. 2015). The western Alps have been recolonised by wolves from the Italian population which had experienced a bottleneck and was reduced to about 100 individuals in the 1970s (Zimen & Boitani 1975). Italian wolves are morphologically and genetically distinct from other wolves (Fig. 4.19). The recolonisation of the Eastern Alps is not as advanced as in the Western Alps. Pioneers in the Eastern Alps came from various source populations. The Alps will become a melting pot of various European wolf populations, enhancing the genetic diversity of the overall Alpine population. One of the challenges regarding the conservation of the isolated Italian population in the past and present is the hybridization with dogs (Boitani 1983 in Boitani 2003).

Expert groups have recommended scientific robust procedures for monitoring lynx and wolf and for an Alps-wide cooperation that can be applied in each country and adapted to national or regional requirements. A strict classification of observations helps to increase comparability and certainty. Wherever new population nuclei emerge, a reliable monitoring system needs to be established. The Alpine lynx and the Alpine wolf population are assessed as Endangered. For both species, the low acceptance leads to conflicts, which are difficult to mitigate and often lead to illegal killing. Anthropogenic losses can strongly affect small populations. Inbreeding is probably the most serious long-term threat to the survival of the lynx population(s) in the Alps. This is not the case for the wolf, which has a sufficient genetic diversity. The wolf shows increasing tendencies almost everywhere, the lynx is stagnant at best. Management structures are an important issue for lynx and wolf.

5. Since 1871, the resident human population in the Alps has almost doubled, from 7.8 million to 15.2 million people (Bätzing 2015). However, the population development varied considerably within the Alps and the population distribution became much more uneven (Fig. 5.1): the majority of people live below 500 m. Areas along major transport routes became urbanised and cities at the edges of the Alps became "commuter towns" for the metropolises surrounding the Alps. Tourist destinations have grown, too. The population increased especially in the western parts of the eastern Alps. The population in higher elevation areas has decreased, mostly because agriculture became unprofitable due to limited mechanisation. The population decrease was most prominent in the Italian Alps (except South Tyrol), eastern Austrian Alps, and some regions in the French Alps. Young people and families moved away, and the population in these communities is considerably over-aged. A further population decrease is expected in areas with unfavourable economic conditions.

Tourism in the Alps has been stagnating on a high level since the early 1980s. About 60 million people visit the Alps every year for daytrips and an additional 60 million people stay for 370 million nights in the Alps every year (Siegrist 1998). However, tourism is spread unevenly across the seasons and across the Alps (37% of municipalities in the Alps offer no tourist beds at all; Price et al. 2011). The influence of tourism on large carnivores and wildlife in general is twofold: Firstly, tourism requires infrastructures (e.g. transport infrastructure, ski slopes, or golf courses), which influences the landscape and the habitat of wildlife. Secondly, touristic activities (e.g. hiking, skiing, paragliding, but also added traffic from visitors) create disturbances for the local wildlife. Nonetheless, the populations of ungulates have increased throughout time. Large carnivores have a high capacity to adapt to human activities. Wildlife and especially large carnivores also represent a chance for tourism as visitors see them as the embodiment of pure nature and untamed wilderness. Wildlife tourism is however weakly developed in the Alps.

Forests in the Alps have been strongly overexploited in the 18th/19th century, but have recovered and forested areas have expanded again in in the 20th century (Fig. 5.5). About 52% of the Alpine area is forested, and "forest creation and management" contributes the majority to recent changes in land cover (EEA 2010).

The fragmentation of the landscape is increasing, which has various negative effects on wildlife (e.g. barrier effect, loss of habitat, increased numbers of traffic collisions). The patch size of most forested areas in EU-countries is smaller than 10 km^2 (EEA 2003). The effective mesh size in Switzerland has decreased to a third of the value of 1885. However, although the value has decreased by the same proportion in the Alps down to about 300–400 km^2, it is still about 10–15 times bigger than the value in the lowlands (Jaeger et al. 2007). The Alps still feature some of the largest unfragmented low-traffic areas (UFAs) in Central Europe, e.g. 8 out of 10 UFAs larger than 200 km^2 in Bavaria are found within the Alps (Esswein & Schwarz-von Raumer 2006), but valley floors can be just as heavily settled and fragmented as the lowlands surrounding the Alps and present considerable barriers to animal movements (Fig. 5.10).

The existence of a sufficient prey base is a key factor determining the successful return of large carnivores (Breitenmoser 1997). Populations of all wild ungulate species have been increasing over the past decades and continue to do so in many Alpine regions except for the chamois. Some countries make regular records of wild ungulate population sizes available (but often do no state census methods clearly), but others like Austria do not (Reimoser & Reimoser 2010). Hunting bag data were the only data sets almost consistently available across the Alps. Of course, data on hunting bags show clear weaknesses, e.g. not being linked with the real hunting effort invested. Nevertheless the data were used to show the development of large herbivores and differences between the Alpine countries.

Red deer have naturally recolonised the Alps, helped by reintroductions. Numbers are still increasing across the Alps according to censuses and hunting bags. Red deer populations in France had strongly declined in the nineteenth century due to loss of habitat and over-hunting, but increased in the 20th century, e.g. from some 5,000 in 1973 to over 39,000 in 2004, reaching relatively high abundance also in the departments in the Alps. Census estimates for the Italian Alpine regions were at 49,074 in 2005 (Carnevali et al. 2009). Red deer went extinct in Switzerland in the 1850s. Natural recolonisation from Austria, supported by releases in different areas let the red deer population recover (Fig. 5.14). Red deer hunting bags in Liechtenstein increased from 185 to 218 between 1993 and 2013. The German Alps are a typical and high-density red deer zone. Austria shows, as compared to the other countries, a strong increase in red deer hunting bags (Fig. 5.18a, b, c). The population size of red deer in Slovenia in 2010 was estimated to be between 10,000 and 14,000 individuals (Adamic & Jerina 2010) and is increasing in numbers and spatial distribution (M. Jonozovič, pers. comm.).

Roe deer are abundant and widespread across the Alps. Hunting bags in the Alpine countries are either stable or increasing. In the French Alps the hunting bag of roe deer increased until the early 2000s and then stabilised (Fig. 5.19b). In 2005, the roe deer population in the Italian Alps was estimated at 184,260 individuals (Carnevali et al. 2009). The roe deer is the most abundant wild ungulate in Switzerland, occurring up to the timberline with a stable population which is estimated at 138,452 individuals in 2013 (BAFU 2014). In Liechtenstein, roe deer populations are low and appear relatively stable with hunting bag numbers of 217 in 1993 and 202 in 2013. Roe deer hunting bags in Austria and Bavaria show a distinct increase compared to the other Alpine countries (Fig. 5.22a, b, c). Roe deer are the most abundant wild ungulate species in Slovenia, with a stable population and hunting bags of over 41,000 individuals in some years (M. Jonozovič, Slovenia Forest Service, pers.comm.).

About 60,000 chamois are currently present in the French Alps (Maillard et al. 2010) following a well-managed population intervention through a controlled harvest (Fig. 5.23). Census data for Italy in 2010 show that there are about 131,714 individuals (Raganella Pelliccioni et al. 2013) with one-third of the population found in the Trentino-Alto Adige region (Carnevali et al. 2009). After a long-lasting increase, hunting bag numbers of chamois are decreasing in Austria and Switzerland since the early 1990s. Data for chamois hunting bags in Liechtenstein indicate harvests of 109 in 1993 and 152 in 2013. Chamois harvests in Germany increased to about 4,000 individuals annually until 2010 (Wotschikowsky 2010). In 2005, the hunting bag data for Slovenia included 2,506 chamois (Adamic & Jerina 2010). The population in Slovenia has a slightly decreasing trend (M. Jonozovič, Slovenia Forest Service, pers. comm.).

Wild boar hunting bag numbers are fluctuating strongly in some of the countries. Wild boar are widespread across France (Fig. 5.30); the hunting bag increased 12-fold from 36,429 in 1973 to 443,578 in 2004 (Maillard et al. 2010). Hunting and culling data for the 2004–2005 season in Italy was at 34,027 individuals (Carnevali et al. 2009). The wild boar harvest is fluctuating in Switzerland (Fig. 5.33): 9,940 individuals were hunted in 2012 and 5,740 in 2013 (BAFU 2014). There is no wild boar population in Liechtenstein. The occasional dispersers are hunted and not allowed to establish a population (W. Kersting, pers. comm.). Despite occasional decreases, the wild boar population has increased steadily over the years in Bavaria (Fig. 5.34), but are not abundant in the Alpine region. Wild boar populations are increasing in Austria based on evidence of increased hunting bags (Fig 5.35; Reimoser & Reimoser 2010). The wild boar population in Slovenia almost tripled in the last 20 years and is still increasing in numbers and spatial distribution (M. Jonozovič, pers. comm.).

Sheep are the most important and most abundant domestic victims of predators in the Alps (Kaczensky 1996). Numbers of sheep (data for whole countries) are increasing in Austria and Liechtenstein. In France and Germany, numbers of sheep have approximately halved since the early and late 1980s respectively. The decrease in Switzerland and Slovenia started in the late 2000s. Statistics from Italy show a sudden drop in numbers in 1999/2000, but an otherwise rather stable sheep population.

The main prey of lynx in Europe are small to medium-sized ungulates (Nowicki 1997), which are more or less of the same size as the predator. Wherever roe deer are abundant, they form the staple food, followed by chamois in the Alps and reindeer in the northern countries. Main prey species of lynx in the French Alps are roe deer and chamois. In the eastern Italian Alps lynx predated on red deer due to its high abundance (Molinari 1998). Long-term studies using radio-telemetry in Switzerland showed that 90% of the lynx diet is comprised of roe deer and chamois (Breitenmoser et al. 2010; Fig. 5.40a–c). Between 1973 and 2013, 2052 sheep and 219 goats were confirmed to have been predated on by lynx in the whole of Switzerland. There is no lynx population in the German parts of the Alps, but a number of unconfirmed kills were recorded in the German Alps between 2005 and 2009 (Wölfl & Wölfl 2011). After the reintroduction of lynx in Austria in the mid-1970s, Gossow & Honsig-Erlenburg (1985) reported a preference for red deer in lynx diet in the eastern Austrian Alps in areas with a very high red deer abundance. Lynx in the Kalkalpen preyed mostly on roe deer, followed by chamois. Studies in the Dinaric Mountains of Slovenia and Croatia showed that roe deer represented 79% of all consumed biomass in the lynx diet (Krofel et al. 2011). A total of 317 depredation cases have been recorded in the Slovenian Alps since 1994, but none since 2011 (M. Jonozovič, pers. comm.). Livestock depredation is generally low due to the low density of lynx and ample availability of wild prey (Kaczensky et al. 2013d).

Under certain conditions, depending e.g. on the population status of predator and prey, on the recolonisation state (e.g. immigrating predators), and on other important mortality factors (e.g. winter mortality or human-made mortality), lynx can have a significant impact on a local roe deer population. The observed roe deer mortality caused by lynx predation (based on the assumed local abundance) varied from 9–63% (Breitenmoser et al. 2007, 2010). Periods of high predation impact triggered severe controversies with hunters resulting in more illegal killings of lynx.

Numerous studies in Europe found that wolves in general preferred to prey on wild ungulates, especially cervids (Bassi et al. 2012). The abundance of red deer and the strong positive response of wolves to red deer density determine the proportions of other species in the wolves’ diet (Jędrzejewski et al. 2000). The seasonal variation in the wolves' diet is particular, with cervids being preyed on both in summer and winter, and livestock becoming more important in some areas in summer. Scat analyses from nine packs from the French Alps showed a relative uniform predation with 76% of wild ungulates, 16% livestock and 8% of smaller prey. Other studies in the French Alps show that chamois, mouflon, roe deer, red deer, ibex, wild boar and domestic sheep were killed in varying proportions depending on seasonality and availability (Fig. 5.47, 5.48; Duchamp et al. 2012). The attacks on livestock in France increased over the years (Fig. 5.50) from 36 sheep killed in 1993 to 8,226 in 2014 (Anonymous 1996, Duriez et al. 2010, DREAL & DRAAF Rhône-Alpes 2011, DREAL 2015). A study in the Piedmont Region of the Italian Alps using a sample size of 2,586 scats showed that in summer, 69.5% of the wolf diet consisted of wild ungulates and 31.9% of livestock (Regine 2008). Palmegiani et al. (2013) found that the wolf's diet in summer comprised mainly of chamois while in winter chamois and roe deer were taken in similar ratios (Fig. 5.51). In 2011, in the Piedmont region, 383 domestic animals, mostly sheep and goat, were killed by wolves (Kaczensky et al. 2013d; Chapter 6.3.2). Wild ungulates made up 65% of the wolves' diet in Switzerland (Weber & Hofer 2010). Red deer constituted the main prey with 32% (frequency of occurrence; Fig. 5.53). Roe deer occurred in 21% of the scat samples. Livestock losses started in 1995 and reached 238 in 2013. No wolf or wolf kills were reported from Liechtenstein up to now. Wolves have not yet permanently settled in the Bavarian Alps, but occur further north in Germany, where wild ungulates constitute almost 100% of prey biomass (Holzapfel et al. 2011). Livestock predation in Germany is relatively low compared to other European countries (Kaczensky et al. 2013d). No wolves are permanently living in Austria, but in 2011, 14 sheep and goats and one calf were killed or wounded (Kaczensky & Rauer 2013). The main prey base of wolves in Slovenia includes red and roe deer, wild boar and chamois (SloWolf 2014). Wolf predation has been found to have a high impact on the red deer population in Slovenia (Kavčič et al. 2011). Domestic animals constitute about 10% of the wolf's consumed biomass (Krofel & Kos 2010, van Liere et al. 2013).

6. All countries in the Alpine arc are signatories to the Bern Convention and the recommendations are implemented in their management plans (Trouwborst 2010). All large carnivores in Europe are covered under Annex II and IV of the Habitats Directive except (in the Alps) for the non-EU countries Switzerland and Liechtenstein (Linnell et al. 2008).

In 2003, a Pan-Alpine Conservation Strategy PACS for the lynx was created (Molinari-Jobin et al. 2003). The strategy was elaborated by the SCALP expert group (Chapter 4.1) and proposed standards aimed at boosting transboundary activities and co-operation from local to international levels. In 2006, the Ministries of Environment of Italy, France and Switzerland signed an "italo-franco-suisse collaborative protocol for the management of wolf in the Alps" (Ministerio dell'Ambiente e della Tutela del Territorio et al. 2006). This protocol takes into account aspects of the Habitats Directive and Bern Convention (Chapter 6.1) as well as the existing national management plans with a common goal of re-establishing and protecting a viable wolf population in the Alpine arc.

Switzerland is the only Alpine country that has a lynx management plan (BUWAL 2004). Germany has an unpublished framework document by the Federal Agency for Nature Conservation (BfN) for lynx management (BfN 2010, Kaczensky et al. 2013b). Bavaria is the only place which has developed a regional management plan for lynx (StMUGV 2008).

The development of a national wolf management plan was addressed in all Alpine countries. Wolf management plans were elaborated in France as early as 1993. The Italian Ministry of Environment with technical support of the Istituto Superiore per la Protezione e la Ricerca Ambientale ISPRA has established National Action Plans for brown bear and wolf in Italy (Anonymous 2012). A concept for the management of wolf in Switzerland was developed in 2004 (BUWAL 2004) and revised in 2008 and 2010 (BAFU 2008, 2010). Additionally, some cantons have created cantonal management plans, which mainly define regional conflict mitigation measures and management competences. In Germany, only a general framework on wolf management exists so far on a national level, but a national management plan is under consideration (BfN 2010 unpublished final report; Reinhardt 2013). Several Länder in Germany including Bavaria have developed regional wolf management plans. According to Reinhardt (2013) these plans, although called management plans, mainly deal with regional conflict mitigation and management competences. The Austrian Wolf Management Plan was finalised in 2012 (KOST 2012, Schäfer 2012). Slovenia has a strategic management plan and a five-year action plan (Majić Skrbinšek et al. 2011) is currently being implemented.

Wildlife in the Alpine countries is managed through legal and practical means such as protective laws and selective hunting. In France, wildlife and environmental monitoring are carried out by the *Office National de la Chasse et de la Faune Sauvage* ONCFS. The role of hunting in Italy is primarily to control wild boar, red deer and roe deer populations (Apollonio et al. 2010). Switzerland has licence hunting across the Alpine range, with 41 federal wildlife reserves where hunting is banned (Imesch-Bebié et al. 2010). Ungulate management and hunting practises in Germany are carried out with the objective of reducing and preventing damage to crops and forests. There is a federal hunting law, but the 16 states all have additional regulations (Wotschikowsky 2010). Austria uses the "Reviersystem" similar to the system in Germany; the Austrian "Bundesländer" are responsible for legislation and management of game (Reimoser & Reimoser 2010). The current Slovenian Law on Wildlife and Hunting was adopted in 2004 and controls the wildlife management system in Slovenia (Adamic & Jerina 2010).

Diseases and epizooties are currently not a great threat to the lynx and wolf populations in the Alps. Wild ungulates have been known to be the host and reservoirs of several common infectious diseases in ungulates (Artois 2003). Such diseases were recently spreading within the Alpine ungulate populations and across Europe (Giacometti et al. 1998, Gortazar et al. 2007). Managing domestic livestock and controlling exposure with infected wild ungulates is an important way of reducing the risk of spreading diseases from wild to domestic animals and vice-versa.

With the disappearance of large carnivores from their historical range, the traditional livestock protection methods were also abandoned, resulting in predation on livestock with the recolonisation of the Alpine Arc by wolf and lynx. Loss of livestock to large carnivores is reimbursed by the government or associations (e.g. hunters) in all Alpine countries.

Depredation cases of lynx on livestock in the French Alps are low and in 2011 the compensation cost amounted to about 20,000 € per year. Compensation for losses to lynx predation in Switzerland amounted to 6’500–25’000 CHF per year between 2006 and 2011 (von Arx & Zimmermann 2013). In Germany, only the states of Bavaria and Lower Saxony have provisions for compensation payments for cases of lynx depredation, as the other “Länder” have not had any damages yet (Kaczensky et al. 2013b). In recent years, there have been no cases of livestock depredation in the Kalkalpen NP in Austria. Compensation costs for lynx attacks between 1995 and 2014 varied from 137–13,225 € in Slovenia. The last damages in the Slovenian Alps occurred in 2011 (M. Jonozovič, pers. comm.).

France paid 2.33 million € compensation for 7,484 domestic animals killed by wolves in the Alps in 2014. In 2012, the number of victims was 5,732, summing up to an amount of 1.84 million € for compensation (Table 6.7). In the Piedmont Province of Italy in 2011, 383 cases of livestock depredation. Direct losses were compensated with 72,953 € and indirect losses with 19,703 € (Boitani & Marucco 2013). In the Swiss Alps, 114,000 CHF for 280 animals killed by wolves was paid in 2011, and 48,500 CHF for 135 animals in 2012 (KORA 2014). In 2011, a total of 26,584 € were paid in compensation for livestock depredation cases all over Germany (Reinhardt 2013). No losses occurred in the Alps that year, but a wolf killed 26 sheep in the Bavarian Alps in 2010. Between 2009 and 2011, there were 15–70 cases of livestock damage by wolf in Austria. Actual damage and compensation costs are not available. Up to 26 animals per year were killed in the Slovenian Alps and damage compensation amounted to up to 3,869 €. Damages started to occur in 2006 and are possibly caused by a single wolf (M. Jonozovič, pers. comm.).

Mitigation measures are used to limit the frequency and the impact of attacks. A number of measures to protect livestock from predation have been identified and include: livestock protection dogs, electric fences and guarding by shepherds (Gehring et al. 2010). In France, 12 million € were spent on wolf attack prevention measures in 2014 (J. Transy, pers. comm.).

7. Lynx are in general a forest-dependent species but are able to use other habitat types as long as enough prey and cover is available. Lynx tend to avoid areas of permanent human activities but in good quality habitats they can adapt to human presence and disturbance (Zimmermann 2004). Lynx live solitary except for females with this year's offspring. Male home ranges are on average 137 km^2 and overlap with those of females (76 km^2, Breitenmoser-Würsten et al. 2001). Young lynx start dispersing at an age of 9–11 months. Male lynx disperse on average 31 km, females 19 km (Breitenmoser & Breitenmoser-Würsten 2008). Long-distance dispersal (e.g. 200 km) is rare in the Alps. Subadult lynx establish their home range close to the home ranges of conspecifics as they need the contact to them. Individual lynx can cross barriers but do not tend to disperse very far and intensively (Zimmermann 2004).

Although wolves have a certain preference for forest habitat in Europe, they are habitat generalists and very adaptable, occurring wherever they can find enough food and experience low human impact. They live in highly social packs averaging about 4–6 animals in Europe (Krutal & Rigg 2008, Marucco & McIntire 2010, Caniglia et al. 2014). Wolves live at low densities over large territories of about 50–300 km^2 (WAG 2014, Marucco et al. 2009). Wolves start dispersing at an age between 5 months and 5 years old (Mech & Boitani 2003). Mean dispersing distance is 50–100 km (Marucco & Avanzinelli 2012, Caniglia et al. 2014), but long-distance dispersal of up to 1,200 km has been observed (Box 7.2). Dispersal rate between females and males seems to differ. All inferred first-generation migrants to the Alps were males (Fabbri et al. 2007).

The various habitat models agree that suitable lynx habitat is still available in large amounts widespread all over the Alps but they disagree whether the eastern or the western Alps contain more highly suitable habitat. Less suitable areas are mainly found at very high elevations (Fig. 7.7; Zimmermann 2004, Signer 2010, Becker 2013). The models showed a preference of forest areas by lynx, followed by shrubs and herbaceous vegetation and an avoidance of intensive agricultural areas. Although lynx presence was negatively correlated with the frequency of urban areas, no negative correlation with distance to roads could be found, indicating that lynx tend to avoid areas with high human activity, but are able to adapt to humans up to a certain degree in presence of good habitat. However, mainly highways seem to negatively affect lynx occurrence (Zimmermann 2004, Schadt et al. 2002a).

The lynx population in the Alps is currently divided into several relatively small genetically isolated subpopulations. Major barriers (e.g. major highways, rivers and high elevation areas) divide the suitable habitat in the Alps into 37 (Fig. 7.12; Zimmermann 2004) or 32 distinct patches (Fig. 7.13; Becker

2013). Lynx can physically overcome all kinds of obstacles and move through unsuitable habitat but the ability (or will) to traverse barriers differs highly between individuals and major barriers are still constraints to the expansion of the population (Rüdisser 2001, Zimmermann 2004). In theory, the population in the Alps is connected to the ones in the Dinaric Mountains, the Jura Mountains and the one in Bohemia-Bavaria. However, major barriers lie in between. Connectivity is still heavily impeded even though especially adult lynx are able to cross some barriers.

With assumed densities ranging from 1–3 individuals per 100 km^2, the models calculated a lynx abundance of approximately 1,000–3,000 individuals for the suitable habitat area in the Alps (Zimmermann 2004, Becker 2013).

After reintroductions in the Alps, the lynx population has initially expanded rather fast across the north-western Alps until about 1985, but has now stagnated for a long time. Although suitable habitat is available, lynx expansion covers less than 20% of the Alps (Molinari-Jobin et al. 2010b). The social structure of the lynx, its need for contact with conspecifics to establish home ranges, its dispersal properties and the low migration rate between subpopulations makes a natural expansion and recolonisation of the rest of the Alps unlikely. The existing dispersal barriers and human caused mortality add to this assessment (Zimmermann 2004, Becker 2013). For the long-term survival of the lynx in the Alpine range and the conservation of the species, it is crucial to connect the small and genetically isolated subpopulations in the Alps.

The results of the various wolf habitat models are mainly in agreement with each other with regard to the main factors influencing wolf presence and distribution. The Alpine wide wolf habitat models predict still a high amount of suitable habitat available for (re)colonisation (Fig. 7.20; Marucco 2011, Herrmann 2011). Higher suitability is indicated in the eastern and north-eastern Alps than in the western and central-western Alps. Regions with very high elevations are generally indicated as very lowly suitable (Glenz et al. 2001, Herrmann 2011, Marucco 2011). High human density and “disturbance” (roads, settlements) were indicated to negatively impact wolf presence whereas prey abundance and diversity, and forest cover were predicted to have a positive effect (Massolo & Meriggi 1998, Herrmann 2011, Marucco 2011, Falcucci et al. 2013).

No subpopulation of wolves is identified in the Alps but the wolf population in the Alps was considered to be a distinct population unit for practical reasons. High connectivity is expected for wolf habitat in the Alps. Depending on the threshold value, only around 70% or 25% of the Alpine range is classified as wolf core areas. Lower connectivity was indicated in the western-central Alps compared to the higher connectivity in the eastern Alps (Fig. 7.31a, b). The population in the Alps is connected to the ones in the Apennines, Dinaric Mountains and Carpathian Mountains. Wolves are able to cross large areas of unsuitable habitat as well as major anthropogenic barriers (Ciucci et al. 2009, Marucco 2011, Falcucci et al. 2013). Nonetheless, high road density can result in significant mortality and reduced habitat quality through fragmentation or by providing easy access to wildlife areas to people and thus limit pack settlements. Natural and anthropogenic factors such as settlements, lakes and high rock areas can decrease connectivity (Marucco 2011).

Herrmann (2011) estimated a potential number of 1,200–1,580 wolves in the Alps based on his habitat model and an assumed density of 1.3–1.7 wolves per 100 km^2. Such estimation is highly speculative, as it does not consider local differences in prey availability but it provides a rough and rather conservative guess.

Under favourable ecological conditions and without persecution, wolf have a high potential growth rate and show high colonisation ability, but their population can drop rapidly under high killing rates

(Chapron et al. 2003). The (re-) colonisation process of wolves starts with the arrival of (mostly male) individual dispersers. It can then take years until the first pack is formed. It is expected that the Alps will mostly be recolonised from the West to the East, even though some dispersers from the Carpathians and Dinarics have been detected in Austria. Marucco & McIntire (2010) predicted an increase in wolf pack density until 2023 in the western Alps, but no big changes in the eastern Alps. However, the rather newly established breeding pairs in eastern Switzerland, in the Veneto (Italy) and in the Slovenian Alps can largely change the dynamics and the speed of the recolonization process and could now be the beginning of the colonisation of the central and eastern Alps.

All habitat suitability models agree that there is good quality habitat for lynx (100,000 km) and for wolf (93,000 km) available in the Alps to host a large population for both species (potential population: 1,000–3,000 lynx, 1,200–1,500 wolves). Major barriers are impeding the population expansion and may result in future genetically distinct subpopulations. The expansion of the lynx population is very slow. Population fluctuations in the north-western Alps did not result in further expansion. Barriers are likely fragmenting the Alps into more than 20 lynx subpopulations (Becker 2013). Low migration rate does hamper the colonisation of still available suitable habitat. Through creation of local population nuclei colonisation and connectivity could be facilitated.

Wolves are highly adaptable and can live almost anywhere. Potential wolf numbers are speculative but indicate that future Alpine wolf population would be demographically and genetically viable. Future wolf presence will likely mainly be defined by human tolerance for the species. Wolves can cross almost any barrier. Wolf pairs can be formed and new population nuclei be started at any site of the Alps. There is no justification to distinguish wolf subpopulations but it could be practical to distinguish several transboundary wolf regions or compartments which will be relevant for management reasons.

To enhance the prediction power of habitat modelling a combination of several methodological approaches was recommended. To ask more precise questions and produce more specific models however, more detailed input data would be needed.

8. Without any doubt, the Alps are offering good ecological conditions for lynx and wolves and provide a high carrying capacity. The Alps can easily sustain demographically and genetically viable populations of the two species (if the inbreeding of the lynx populations is remedied). The main challenges for the survival of lynx and wolves in the Alps are not ecological factors, but the coexistence with local people and land users. Three main aspects of conflicts were identified:

First, the fear people have of these large carnivores. Although wolves or lynx are not dangerous animals, the concerns of local people have to be taken seriously, especially because large carnivores can easily get used to human presence and activities. Above all, habituation (e.g. through making food available) has to be avoided, and protocols have to be developed how to deal with habituated large carnivores.

Second, the return of wolf and lynx requires major adaptations in the livestock husbandry practices, mainly with regard to free-ranging sheep. Attacks and losses by and through large carnivores are addressed through (1) preventive measures, (2) paying compensation for losses, and (3) the eventual removal of notorious stock raiders in some of the Alpine countries. However, the changes in sheep husbandry to coexist mainly with wolves are fundamental and should also be considered in a wider context, e.g. conserving alpine habitats and maintaining economically viable agriculture in the Alps.

Third, the presence of lynx and wolves requires adaptations in the wildlife management systems. The predation impact of the large carnivores on wild prey population has not yet been exhaustively studied in the Alps, but the competition of the large carnivores with human hunters is – besides the attacks of wolves on livestock – the most important source of conflict. Experiences with lynx and wolf predations come mainly from the western part of the Alps, where large game densities generally are lower. In the eastern and north-eastern Alps with very high wild ungulate abundance, hunting has also a different tradition and a much higher economic importance. The reintegration of large carnivores into the wildlife management system will require an intensive and lasting discussion between wildlife managers, hunters, and foresters.

1. Introduction

The renaissance of large carnivores – brown bear *Ursus arctos*, wolf *Canis lupus*, and Eurasian lynx *Lynx lynx* – in the Alps is a considerable challenge for every Alpine country. On the one hand, these species are protected by national laws and international treaties, on the other hand, their return causes severe controversy, e.g. with regard to livestock husbandry or hunting and wildlife management practices, which are no longer adapted to the presence of large carnivores, but also with regard to local people who have lost the tradition to live with these species often considered to be dangerous. Society's determination to conserve these species (exemplified by their legal protection) often collides directly with the need for interventions to solve problems at local level.

For some people, large carnivores represent the archetypes of pure nature and wilderness. This is clearly not so; these animals also recolonise human-dominated landscapes and have a high potential to adapt to human presence and activities. However, vast mountain ranges such as the Alps provide extended and rather well connected complexes of "close to natural" habitats and can therefore host large viable population of megafauna. The European Environment Agency (EEA 2010) analysed a "Wilderness Quality Index" for the whole of Europe. The Index was based on human population density, road density, distance from nearest road and naturalness of land cover. Mountains showed a high correlation with areas of high Wilderness Quality Index and the top 10% wildest areas can be found in high latitude or high altitude areas. About 9% of the Alpine area belongs to the top 10% of wildest areas in Europe (EEA 2010). But the Alps are also the most intensively used mountain range of the world. Wolf and lynx return to a human-made and human-dominated landscape, and protected areas, which cover only about 25% of the Alpine range (ALPARC 2014), are all too small to host viable populations of large carnivores. Consequently, large carnivores will only be able to recover and survive if people are willing to share their living space.

A particular challenge is related to the spatial scale of the land tenure systems of all large carnivores. Pack territories of wolves or home ranges of lynx and bear stretch beyond the average hunting ground, area of supervision or protected area, and viable populations of these species are larger than provinces or even countries. Large carnivore conservation and management consequently requires cooperation at regional and national level. International cooperation implies the definition of common goals and standards and the development of shared management principles allowing reaching the conservation goals at the level of the entire population.

The need for transboundary cooperation between countries sharing populations of wolf, bear, or lynx has been addressed in the *Guidelines for Population Level Management Plans for Large Carnivores* (Linnell et al. 2008), which have been contracted by the European Commission. In the Alps, international cooperation with regard to large carnivore conservation was taken up by the Platform WISO (Wildlife and Society[1]) of the Alpine Convention. The WISO Platform has developed Guidelines "Large Carnivores, Wild Ungulates and Society" (WISO Platform 2011), adopted by the 11th Alpine conference in Brdo, Slovenia, in March 2011. As a general orientation, the WISO Guidelines state: "*Large carnivores and wild ungulates are preserved in balance with their habitat, other wildlife and human interests. Conflicts with human interests are addressed and negative impacts are counterbalanced.*" The WISO Platform was mandated to "Develop practical goals and management options for

[1] For details see http://www.alpconv.org/en/organization/groups/WGCarnivores/default.html.

the recovery and the conservation of wolf, lynx, and (depending on available funding) bear populations in the Alps, and present the conclusions to the relevant institutions of the Alpine Convention[2]".

In order to give a well-founded background to the WISO Platform and other interested institutions, the RowAlps (Recovery of Wildlife in the Alps) Project was launched with the goal to develop and present practical conservation and management options for wolf and lynx in the Alps. The project addresses three Objectives, namely (1) to compile relevant biological, ecological and wildlife management information, (2) to review human dimension aspects and factors influencing people's tolerance towards large carnivores, and (3) to translate these findings into practical management options or scenarios. While the first two tasks were tackled by experts in the respective fields, the third objective is taken care of by a working group consisting of representatives or mandated experts of governmental agencies involved in wildlife conservation and management in all Alpine countries.

The Goal of Objective 1 was:

"***To review and assess, based on available scientific publications and reports, statistical materials and up-to-date experience, the present situation of wolf, lynx and their prey populations in the Alps, the expected development of the populations, and discuss challenges in wildlife management as a consequence of the return of the large carnivores***".

It was not the aim of our work to provide new data or models for wolf and lynx in the Alps, but rather to present a broad overview on topics relevant for the conservation of the two species. All management options need to be implemented in the context of the local socio-economic realities and regional or national wildlife management systems, and these differ considerably within the Alps and between Alpine countries. When discussing management options, such differences need to be understood and considered as much as local differences of the status of predator and prey.

Another important aspect is time. The large carnivores disappeared from the Alps (with exception of a tiny population of brown bear in the Trentino, Italy) in the second half of the 19th century. At that time, wild ungulates were already rare or totally eradicated. In the 20th century, the large herbivores started to recover, and this process is still ongoing in some areas of the Alps. The return of large carnivores can be seen as the last step in the re-establishment of the Alpine megafauna. The present return of the wolf seems to be a fast and very dynamic process, as individuals can travel fast and far and can nowadays reach almost every corner of Europe, including countries like Denmark or the Netherlands, which do not seem to be prime wolf lands. But the spread of the "population" (a group of permanently resident animals with reproduction) and the establishment of viable populations will however take much longer and last for several decades. The return of the large carnivores is hence a phenomenon that calls for immediate solutions as well as long-term concepts. The world may look different 50 years from now than what it looked some 100 years ago when the large carnivores vanished. Wolf and lynx disappeared from the Alps during the second half of the 19th century. Thus, the end of the 19th century is our reference point when comparing the present situation with the phase of their eradication.

[2] Translation from the mandate of the WISO Platform for the years 2013–2014, available in German, French, Italian, and Slovenian at: http://www.alpconv.org/en/organization/groups/WGCarnivores/default.html.

2. Methodological remarks and geographic scope

2.1. Approach and methods

We compiled information on the two target species wolf and lynx, but also on their main prey (red deer *Cervus elaphus*, roe deer *Capreolus capreolus*, chamois *Rupicapra rupicapra*, and wild boar *Sus scrofa*) and the principle domestic prey, the sheep. Prey abundance and habitat quality provide the environmental parameters defining the potential (e.g. carrying capacity) populations of wolves and lynx in the Alps. The lynx' preferred living space in Europe are forests. Wolves are more plastic and can live in a variety of habitats, but the experience with the recolonisation of Western and Central Europe so far have also shown a preference for forested landscapes. As an indicator of habitat quality across the Alps, we therefore focus on the development of the forested areas. Potential threats or limiting factors to the presence or survival of large carnivores in cultivated landscapes compared to natural areas are anthropogenic constraints or human-induced mortalities as a consequence of habitat fragmentation (e.g. through traffic infrastructure), competition for space or resources (e.g. prey) and conflicts leading directly or indirectly to increased mortalities. We have compiled information on all parameters that we consider to have a potential impact at the level of the populations, hence obstructing the expansion, limiting the abundance, or restricting the connection between (sub-) populations.

The prime sources of information were (scientific) publications or reports, followed by materials available on websites and data provided by colleagues. Our research revealed that information on large carnivores is more readily available than on prey or habitat. Data on wild ungulates are amazingly cryptic, considering the importance of hunting in all Alpine countries. Data on wildlife are often not published in annual statistics and not available through the internet, and statistics available on local, regional and national level often do not match. Furthermore, there is no consistency in the compilation of data between the different countries, and often not even between provinces within one country. It was not possible to assess or check the data within the present project; we have compiled and merged them to the best of our capacity, and we have, where complete datasets were not available, taken local or regional data (often gathered in the frame of a scientific project) that we consider to be representative for the general situation or which at least illustrates a certain important trait. However, there are a lot of gaps and need for further research. As we cannot present all information compiled in this report, we make all data sets which are not copyright restricted available through the MALME website (Appendix I; http://www.kora.ch/malme/; also accessible via www.kora.ch). We encourage all interested readers to make use of the information provided there!

2.2. Geographic scope

The geographic scope of the work is the Alpine region as described in the Annex to the Alpine Convention (Fig. 2.1; PSAC 2010a). Wherever possible, statistics presented in this report refer to this perimeter. If data were not available for the Alps as defined by the Convention, we used the administrative units providing the nearest match. If such an approximation differs significantly from the Alpine Convention area, it is mentioned in the text. Large carnivores will anyway not care about the definitions of the Convention. The "Alpine wolf population" or the "Alpine lynx population" are not geographically strictly defined populations, but rather terms allowing distinguishing e.g. "wolves living in the Alps" from conspecifics living in the Apennine or in the Dinaric Range. For the large carnivore populations, we use the definitions and delineations as proposed by Kaczensky et al. (2013a).

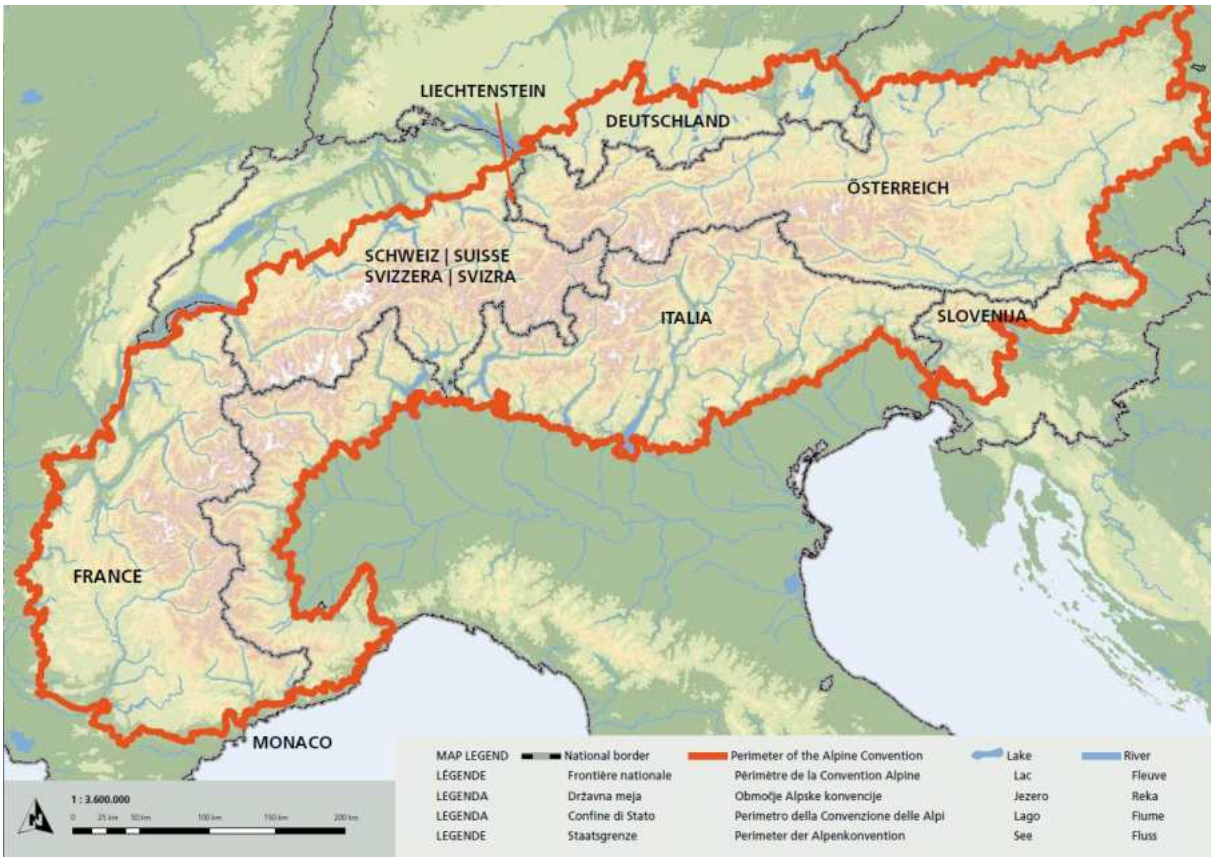


Fig. 2.1. Perimeter of the Alps as defined by the Alpine Convention. The perimeter encompasses 190,959 km² and eight countries: France, Monaco, Italy, Switzerland, Liechtenstein, Germany, Austria, and Slovenia. Monaco, though a signature country of the Convention, was not considered in the report as it has no habitat for large carnivores.

3. Return of lynx and wolf to the Alps

3.1. Why do the large carnivores return to the Alps?

3.1.1. Eradication of lynx and wolf in the Alpine countries

The historic decline and eventual eradication of wolf and lynx (and bear) proceeded in parallel and was related to the expanding human population and agricultural areas. Large predators were perceived as threat to livestock and game animals and governments paid bounties for their persecution. This was however complemented by a massive destruction of forests to gain land for agriculture (Breitenmoser 1998a) and a steady decline in wild ungulates, accelerated in the 18^{th} and 19^{th} centuries (Breitenmoser & Breitenmoser-Würsten 2008). Increasing numbers of sheep, goats, cattle and horses affected the forests negatively due to browsing, and out-competed the wild ungulates. The large predators were forced to kill livestock, thus provoking and facilitating their persecution. At the same time, there was a spread of firearms and hunger which supported not only the extermination of large carnivore but also of large herbivore species. In Central Europe, red deer was practically extinct, roe deer very rare (Zimen 1978). In Switzerland, only chamois survived in low numbers. Ibex, red deer, roe deer and wild boar had gone (Breitenmoser 1998a).

Lynx

Lynx went first extinct in regions that were most densely settled by humans, like the lowland of western and central Europe, Italy, and the plains of Hungary. As a mandatory hunter and carnivore, lynx suffered more from deforestation and the decline of its natural prey than the other large predators (Breitenmoser 1998a). By 1800 the species only occurred on the major mountain ranges in temperate Europe and boreal forests of Scandinavia and Russia. By 1950 the species was eradicated from central, southern and western Europe. It only survived in the Carpathian Mountains, south-western Balkans and Fennoscandia (Fig. 3.1) but in very limited numbers (Breitenmoser 1998a).

The Alps were the last refuge for lynx in western/central Europe. It had disappeared from the Swiss Plateau and the Jura Mountains by the end of the 18^{th} century. Between 1800 and 1850 it disappeared abruptly from the eastern Alps (Austria, Italy and Switzerland) and survived only in the western Alps until the early 1900s (Fig. 3.2, Table 3.1; Breitenmoser 1998a, Breitenmoser & Breitenmoser-Würsten 2008). The last reliable record of lynx in the Alps came from the Val de Susa (Italy) in 1929 (Breitenmoser & Breitenmoser-Würsten 2008).

Wolf

By 1800 wolf had been eradicated from the British Isles and from the coastal lowlands of France, Benelux countries, Denmark, Germany, and Poland. In the rest of the continent, a continuous wolf population resisted, but was increasingly fragmented during the following 150 years. These small and isolated populations were more vulnerable and became extinct one by one. Up to 1973, large populations remained only in Eastern Europe (Russia and the Carpathian region), and smaller and isolated populations in former Yugoslavia and Greece, in Italy and on the Iberian Peninsula (Fig. 3.3; Zimen 1978).

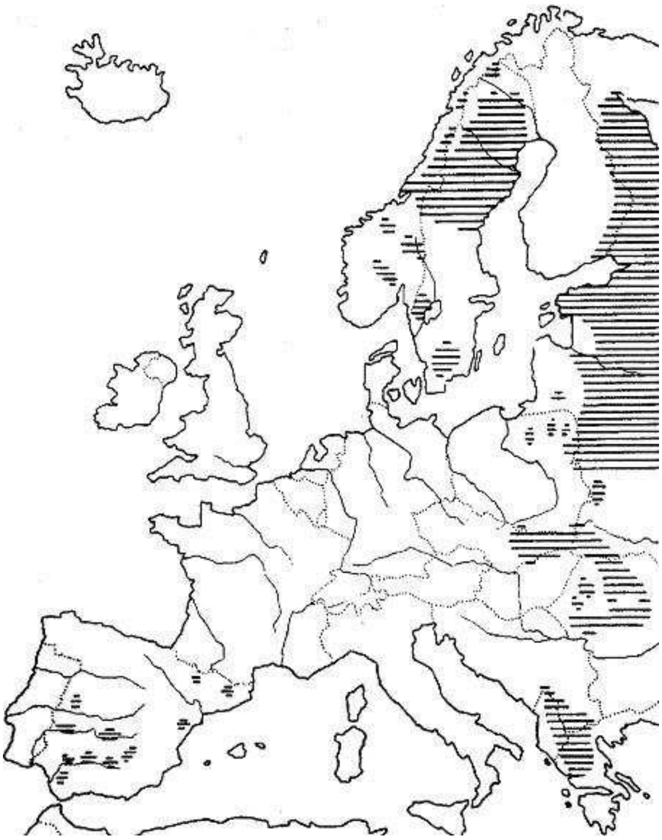


Fig. 3.1. Distribution of the Eurasian lynx (and the Iberian lynx *L. pardinus*) in Europe in the 1960s (Kratochvil et al. 1968).

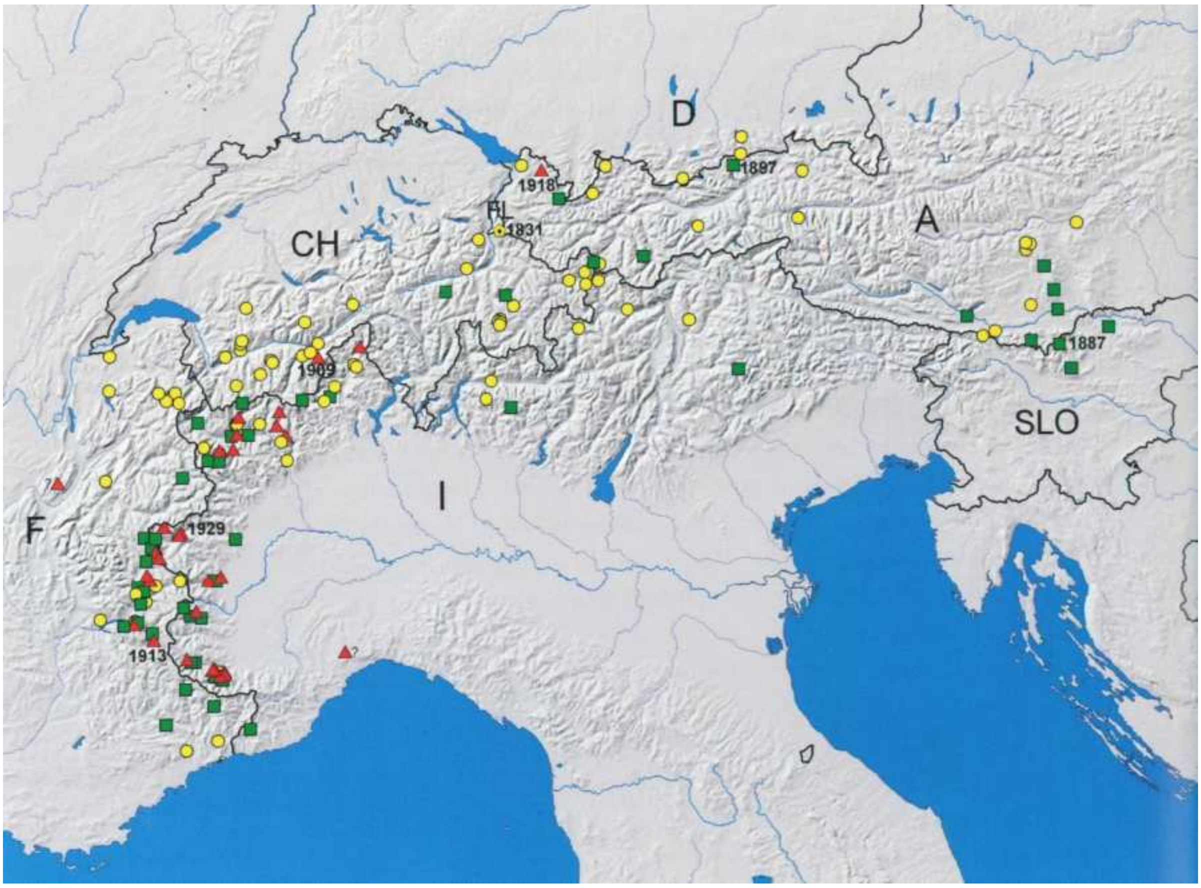


Fig. 3.2. Historical distribution of lynx in the Alps. The map illustrates the decline of lynx in the Alps since the middle of the 19th century. Yellow symbols: 76 records from 1850–1874; green symbols: 48 records from 1875–1899; red symbols: 32 records from 1900–1930. In each of the Alpine countries the last record is indicated by the respective year. Source: Breitenmoser & Breitenmoser-Würsten 2008.

Table 3.1. Year and region of eradication of lynx and wolf in the Alps per country. Sources: Lynx: Kos 1928, Eiberle 1972, Breitenmoser & Breitenmoser-Würsten 2008; Wolf: Zimen 1978, Broggi 1979, de Beaufort 1987, Bauer 1989, Delibes 1990, Etter 1992, Lustrat 1993, Zedrosser 1995, 1996, Erhatič Širnik 2003.

Country	**Eradication of lynx (year & region)**	**Eradication of wolf (year & region)**
France	1913	1898 (1939 in other parts of FR; cf. Fig. 3.4)
Italy	1929, Val de Susa	By the beginning of 1900 it had disappeared from the Alps. In 1973 it was at his lowest in IT ever, but was never eradicated.
Switzerland	1894 last shot, 1909 last seen, both Valais	1872 Ticino
Liechtenstein	1831	1812
Germany	One of the last lynx was killed in 1888 in Oberbayern, not far from Berchtesgaden National Park.	1847 Bavarian Forest, 1891 Saarland
Austria	1897 Tyrol, 1918 Vorarlberg	Last autochthonous presence 1879-82 Styria, however single individuals appeared (and were shot) every now and then afterwards. Thus, Austria was never completely "wolf-free".
Slovenia	1887 (1907 just east of the Alps)	Despite several attempts in time, wolf was never eradicated in Slovenia. In the 2nd half of the 19th century, however, wolf (and bear) only survived in the extensive Snežnik-Kočevje forests (southern Slovenia), which were at that time managed by the proprietors who had strictly prohibited bear hunting in their hunting reserves.

The wolf disappeared from the northern Alps by 1850 and from the southern Alps by 1900 (Table 3.1; Breitenmoser 1998a). Zimen (1978) assumes the longer resistance in the south to be due to the milder climate which allowed keeping livestock on pastures for a longer period of time, sometimes even all year-round. Hunting alone has probably only influenced the number of wolves locally and temporarily. Already in the Middle Ages, battues for killing wolves were organised and bounties were paid (e.g. Alleau 2009 for France, Erhatič Širnik 2003 for Slovenia) without wiping out the species. Only the massive intervention at the level of the landscape and the substantial reduction of wild ungulates led to the final eradication of wolves (Zimen 1978).

At the end of the 18th century almost 90% of France was still occupied by wolves. In 1900 its range was reduced to 16%, 1908 to 4.25% and 1923 to 1% (Fig. 3.4). At the beginning of Wold War II the species was extinct. The last breeding wolves occurred in central-western France (de Beaufort 1987).

The wolf had disappeared from the Italian Alps by 1900, and in 1950 there were no wolves left in Sicily. Distribution dropped drastically between 1945 and 1970, when the wolf population was at its lowest (Fig. 3.5). In 1983, the wolf population seemed to be distributed in four isolated areas in central and southern Italy and a slow recolonisation of former areas started (Delibes 1990).

The eradication of wolf in Switzerland lasted from 1500 to 1850. The species firstly disappeared from the Plateau, soon after from the north-western Alps, then from the eastern Alps, the Jura Mountains and finally from the south-western Alps, namely the Canton of Valais. The last refuges were in the valleys in the Ticino and the Misox, but around 1850, the wolf also disappeared from the southern Swiss Alps within a decade (Etter 1992).

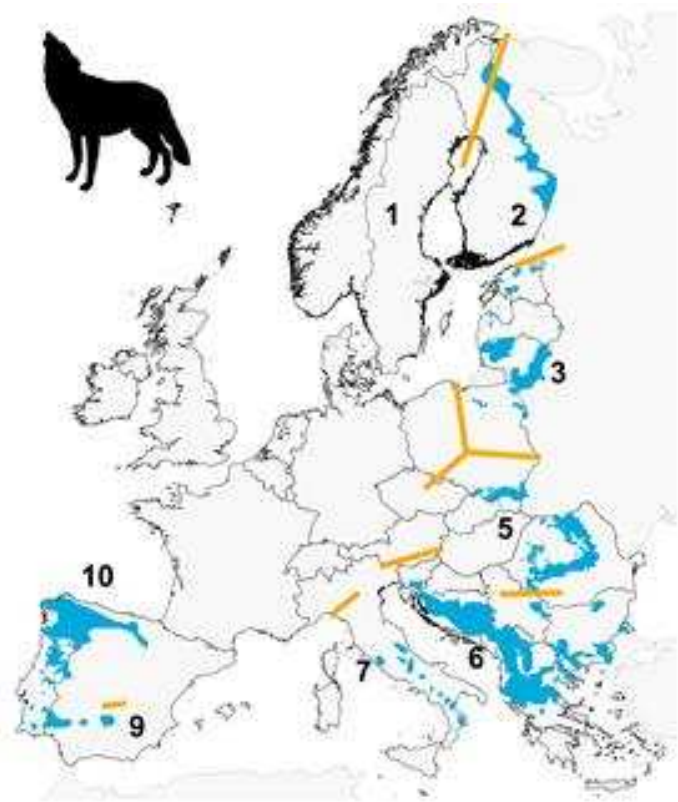


Fig. 3.3. Distribution of wolf in Europe at its lowest extent in the 1950–1970s. Numbers and orange lines refer to populations as defined by the LCIE. Source: Chapron et al. 2014.

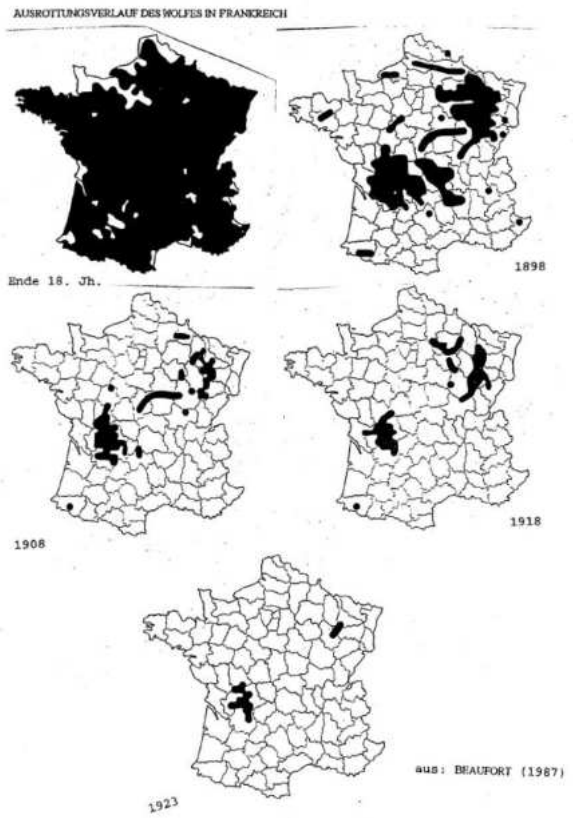


Fig. 3.4. Changes in wolf distribution from the end of the 18th century to 1923. Source: de Beaufort 1987.

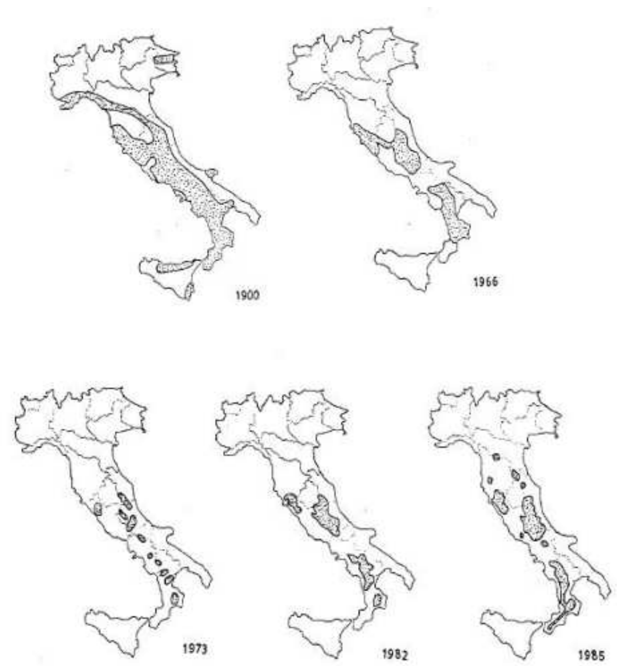


Fig. 3.5. Changes in wolf distribution in Italy from 1900–1985. Source: Delibes 1990.

In the 19th century there were three major retreat areas for wolves in Austria: Carinthia-Styria/Lower Austria, Mühlviertel/Waldviertel, and the border area Upper Austria/Styria. Finally the species disappeared from these regions as well, and all wolves observed in the 20th century in Austria were migrators from the Dinaric or Carpathian populations (Zedrosser 1996).

In all Alpine countries where the species was extinct, namely France, Switzerland and Germany, some long-range dispersing wolves have occasionally showed up since 1950 – and were soon killed (Breitenmoser 1998a).

3.1.2. Reasons for the return of lynx and wolf to the Alps

As the fall of the large predators was a consequence of both human persecution and degradation of their environment, their recovery required a change of human attitude and an improvement of the ecological situation (Breitenmoser 1998a). A complete change in the method of forest exploitation and the growing sensitivity of people for the protection of nature in the first half of the 20th century (which resulted in strict forest protection and restrictive hunting laws in many countries) were the basis for the recovery of the forests (Breitenmoser 1998a). As a consequence the forested areas increased all over the Alps (Chapter 5.2). After they were granted a certain legal protection (at least regulated hunting) and wildlife refuges, the wild ungulates started to recover and expand from remnant source populations, but their renaissance was supported by numerous – and most often not documented – translocations and reintroductions (Chapter 5.3). A swift increase in all wild ungulate populations – which is still continuing for roe deer, red deer and wild boar (Chapter 5.3) – was the result (Breitenmoser 1998a), most prominent after World War II. The ecological recovery was facilitated by industrialisation, which drew people away from rural areas (Chapter 5.1). As a consequence, the number of goats and sheep in the Alps declined drastically in the first half of the 20th century (Chapter 5.4).

Subsequently, both wolf and lynx were granted some legal protection in all Alpine countries. The lynx was legally protected in Liechtenstein as early as 1937, whereas the wolf was listed as a protected species in Germany only in 1990. Many countries in western Europe granted legal protection to large carnivore species only as an endorsement of international treaties such as the Council of Europe's Bern Convention or the EU Habitats Directive (Chapter 6.1). These species were extinct, and until recently, no one could imagine that they would return so soon…

In Slovenia, the Hunting Association decided in 1971 to give more attention to the wolf, and five years later the wolf was granted a closed season. In 1990, the Slovenian Hunting Association declared itself against any form of wolf hunting. At the normative level, however, the wolf was finally protected by the Decree on Protection of Rare and Endangered Animal Species adopted in 1993 (Erhatič Širnik 2003).

The recovery of forests and wild ungulates and the legal protection prepared the ground for the return of the large carnivores for all Alpine countries. The lynx still existed in decent numbers in the Carpathian Mountains, but as a consequence of its rather low colonisation capacity (Chapter 7.1), there was no chance of a spontaneous return of the lynx to the Alps. As early as in the 1950s, some conservation pioneers started to think about its reintroduction (Breitenmoser & Breitenmoser-Würsten 2008). The return of the wolf, on the other hand, was a consequence of the strengthening of the remnant populations in the Apennines, in the Dinaric Range and in Eastern Europe.

3.2. Reintroductions and development of the lynx population in the Alps

3.2.1. Lynx reintroductions and development until 1995

Today, the Alpine population consists of several occurrences, all originating from reintroductions in the 1970s (Table 3.2): Switzerland 1970–76 (Breitenmoser et al. 1998), Slovenia 1973 (Cop & Frkovic 1998) and Austria 1977–79 (Huber & Kaczensky 1998). There were also several attempts to initiate the reintroduction of lynx into the German Alps but none of the projects could be carried out because of very controversial attitudes towards the species and because of competition between institutions (Kaczensky 1998).

Table 3.2. Lynx reintroductions in central and west Europe. Source: von Arx et al. 2009 with data compiled from Breitenmoser et al. 2001, von Arx et al. 2004, Breitenmoser & Breitenmoser-Würsten 2008, Linnell et al. 2009. *m/f = males/females. Sometimes the information is not or only partly available. **Fate: "success" in brackets as these populations survived up to now with reasonable numbers of animals, however their long-term survival is not yet secured, e.g. because of inbreeding.

Population	Location of the reintroduction	Years	Numbers of animals (m/f)*	Origin of animals	Fate**
Bohemian-Bavarian	Bavarian Forest (DE)	1970–74	5–10	3 wild, 2 captive	failed
	Sumava Mts. (CZ)	1982–89	18 (11/7)	wild	(success)
Dinaric	Kocevje (SI)	1973	6 (3/3)	wild	(success)
Alpine	Western Swiss Alps	1971–76	12 (7/5)	wild	(success)
	Engadin (CH)	1972/80	4 (2/2)	wild	failed
	Gran Paradiso NP (IT)	1975	2 (2/0)	wild	failed
	Austrian Alps	1977–79	9 (6/3)	wild	failed
	Eastern Swiss Alps	2001–08	12 (6/6)	wild	uncertain
Alpine/Jura	Swiss Plateau	1989	3	unknown	failed
Jura	Swiss Jura Mts.	1972–75	10 (5/5)	wild	(success)
Vosges-Palatinian	Vosges Mts. (FR)	1983–89	21 (12/9)	19 wild, 2 captive	uncertain
Podyji	Podyji NP (CZ)	1993–94	6 (2/2)	captive	failed
Kampinos	Kampinos NP (PL)	1992–99	31 (14/17)	captive	uncertain
Harz	Harz Mts. (DE)	since 2000	28 (9/15)	captive	(success)

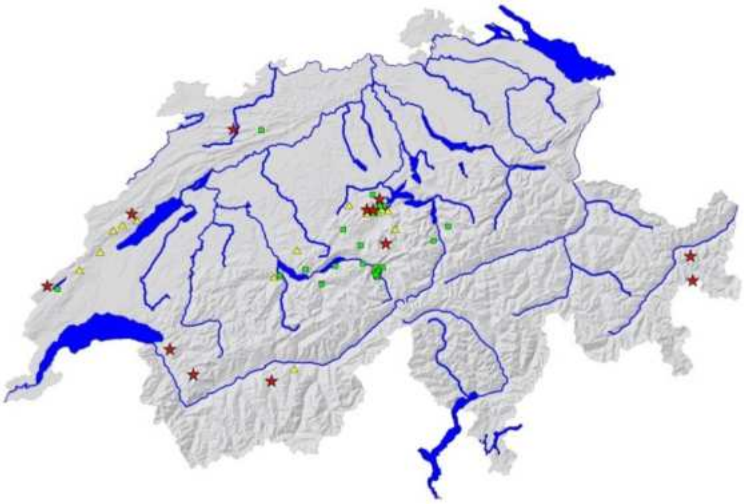
The lynx brought back to the Alps were all Carpathian lynx *L. l. carpathicus*, at that time the geographically nearest autochthonous population. The strongest population is in the north-western Alps of Switzerland. Although lynx immigrated into neighbouring countries (France, Italy; see below) the forty years since the first releases have not allowed establishing a continuous population throughout the Alps. In this Chapter, we describe the development of lynx in the Alps starting from the reintroductions in Switzerland, Slovenia and Austria, to the recolonisation into neighbouring countries in the subsequent years until 1995. The more recent development is in Chapter 3.2.2.

Switzerland. Between 1970 and 1976, at least 14 lynx were released in five different places in the Swiss Alps (Breitenmoser et al. 1998). Furthermore, 8–10 lynx were released in the Jura Mountains and 3 lynx in the Swiss Plains (Fig. 3.6). There is some uncertainty about the exact numbers of lynx set free in Switzerland because all releases were made clandestinely and private actions took place apart from the officially authorised operations. There was some legal uncertainty about the release of wildlife back then. Not all releases were successful, but the reintroduction in the north-western Alps founded a population that covered an area of some 4,000 km² in 1981. From the western Swiss Alps, lynx moved into Italian and French territory (see below). Towards the eastern Swiss Alps, the expansion was slower and ceased in the mid-1980s (Fig. 3.6). There, the area occupied even shrunk, particularly in the former centre of the population (Breitenmoser et al. 1998). The reason for the stagnation was unclear, but Molinari-Jobin et al. (2001) assumed it was due to the natural and artificial barriers that hindered individual lynx dispersal, and, maybe more importantly, illegal killings, which hurt the population demographically. Breitenmoser (1998b) mentioned a limited inherent capacity of lynx to expand their range as another potential factor. By 1995, the population in the Swiss Alps covered an area of about 10,000 km² and, based on size and overlap of average home ranges of radio-tagged lynx, was estimated to include some 50 adult residents (Breitenmoser et al. 1998).

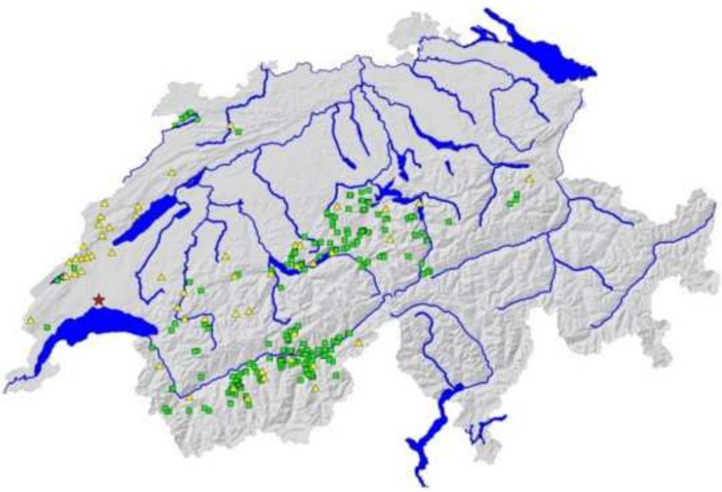
Slovenia. In 1973, 6 lynx (three males, three females) were released in the Kocevje area of Southern Slovenia (Fig. 3.7). The initial population increase and expansion was much stronger than with any other reintroduction. The released lynx met excellent conditions in the release area: extended forests with plenty of prey, but a rather sparse human population (Cop & Frkovic 1998). An important aspect was the high reproduction in the early stage (Potočnik et al. 2009). Particularly the expansion towards south-east into Croatia was very dynamic and the population reached Bosnia-Herzegovina already in 1984, 185 km away from the release site (Cop & Frkovic 1998). The population increased so fast that already in 1978, quota hunting of lynx was legalised in Slovenia. From 1978 to 1995, a total of 229 individuals were hunted. Additional 48 lynx mortalities from other reasons were also known (Cop & Frkovic 1998). In 1984 the first lynx reached the Julian Alps; however the intense harvest hindered the further expansion and recolonisation of the Alps (Fig. 3.7). The hunting season and the quota were subsequently reduced and hunting became restricted to the core area of the population in southern Slovenia (Cop & Frkovic 1998). By 1995, lynx occupied an area of 3,700 km² in Slovenia and an additional 3,000 km² in Croatia. The Hunters' associations estimated the entire population at 300 individuals, whereas the estimate of Cop & Frkovic (1998) was 140 animals.

Austria. Nine wild lynx (six males, three females) were translocated from the Carpathian Mountains to Turrach, Styria between 1977 and 1979 (Huber & Kaczensky 1998). Field projects continued until 1982, when the monitoring of the released animals ceased. Five years after the release lynx had spread as far as 120 km from the site of reintroduction (Fig. 3.8), but observations were few and mostly unconfirmed. There were scattered individuals rather than an established population (Gossow & Honsig-Erlenburg 1986). Observations became scarce during the 1980s, and the reintroduction was believed to have failed, when in 1989 a series of sheep kills assumedly by lynx in Carinthia reactivated the awareness and more observations were reported again. The Carinthian Hunters' Association formed a lynx group in 1992 to collect lynx data. However, most of the information collected was not valid proof of lynx presence. The only reliable continuous observations came from southern Carinthia, where lynx immigrated from Slovenia (Huber & Kaczensky 1998). By 1995 lynx was constantly present only in the Karnische Alpen and in the southern Gailtaler Alpen. All other records were either transient animals or erroneous reports. The very low number of known mortalities compared to the reintroductions in Switzerland and Slovenia as well as the fact that confirmed observations of lynx reproduction were missing let Huber and Kaczensky (1998) conclude that there was no established lynx population in the Austrian Alps.

a) 1971–1981



b) 1982–1992



c) 1993–2003

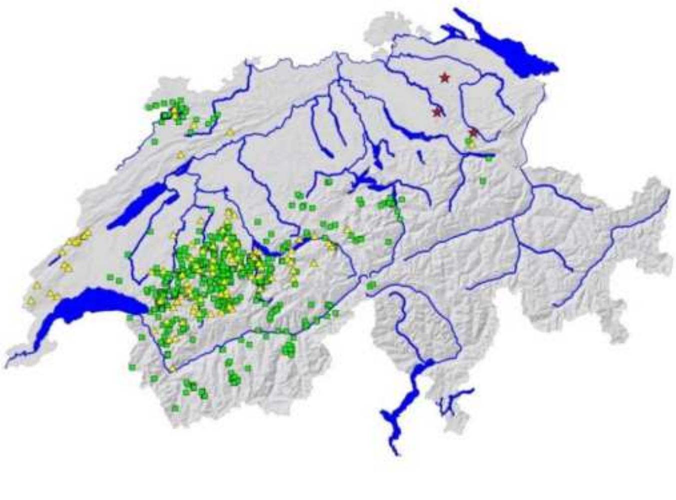


Fig. 3.6. Release sites (red stars) of lynx in Switzerland and development of the populations since 1971. Yellow triangles = lynx found dead, green squares = confirmed livestock killed by lynx. Source: Breitenmoser & Breitenmoser-Würsten 2008 from various sources.

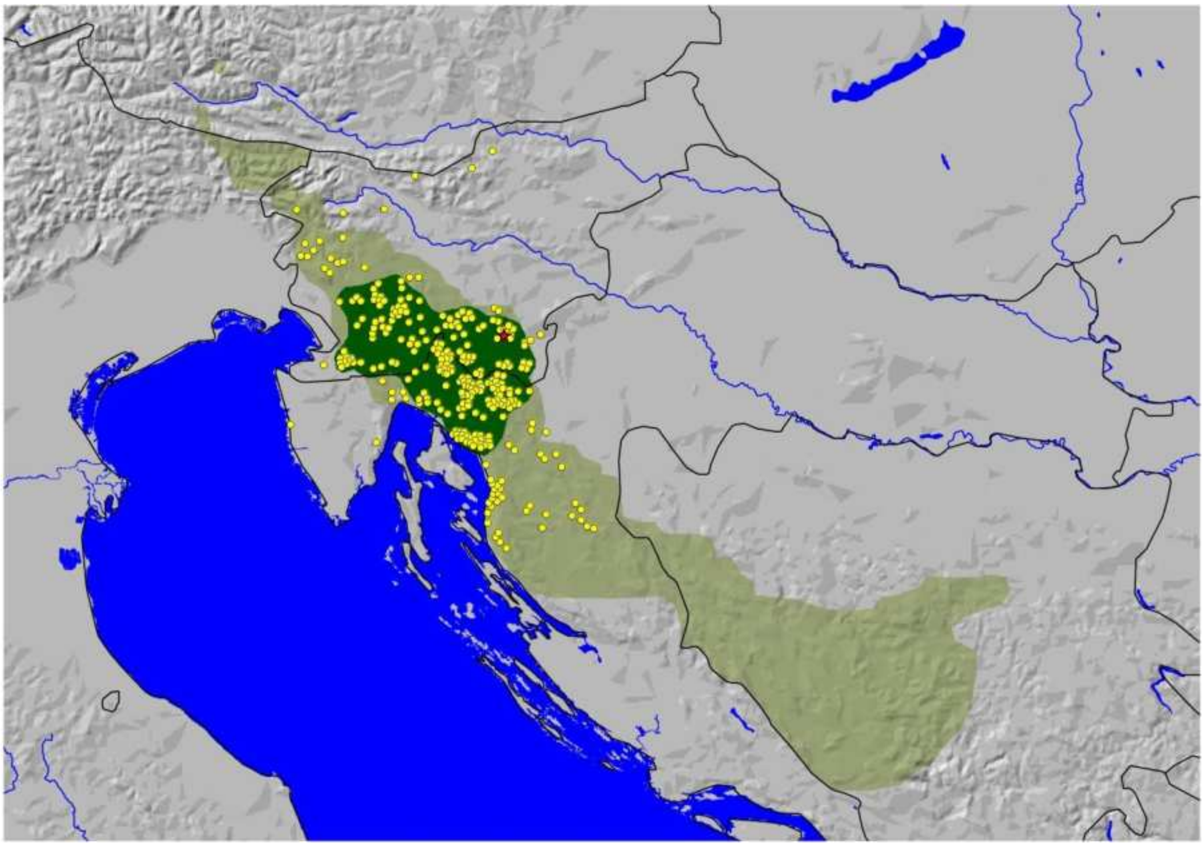


Fig. 3.7. Distribution of the reintroduced lynx in Slovenia and Croatia. The yellow dots (n = 300) indicate the locations of lynx hunted or found dead (130 in Slovenia, 170 in Croatia) between 1978 and 1997. The three north-eastern most shootings in the Karawanken could have been lynx that immigrated from Austria. Dark green is the core area; the red star indicates the release site. Source: Breitenmoser & Breitenmoser-Würsten 2008 from various sources.

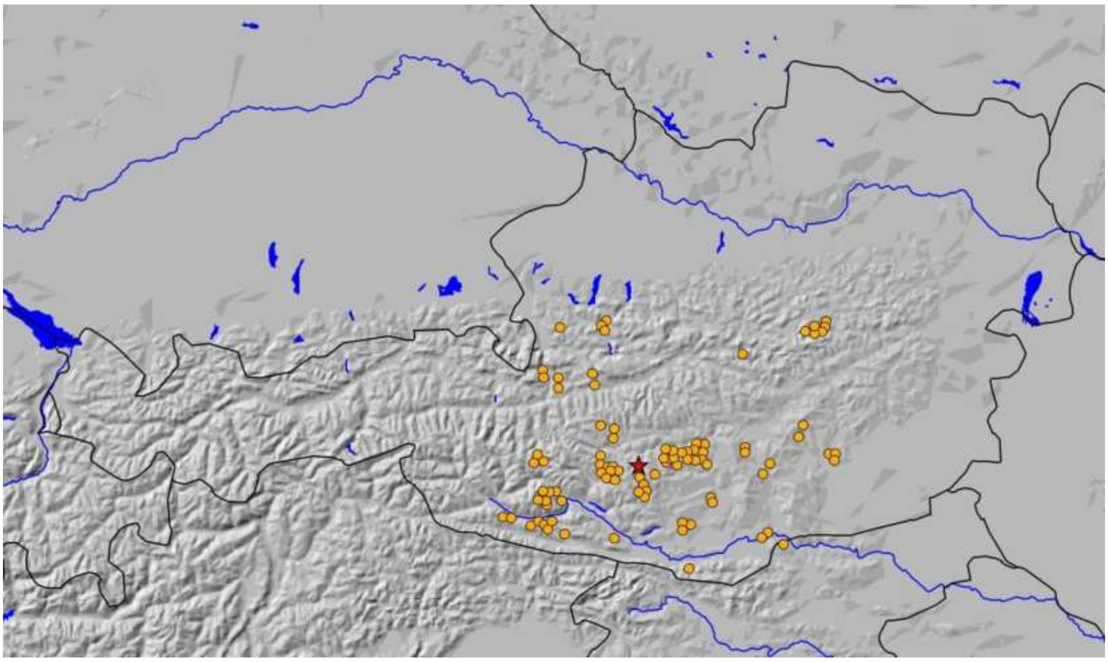


Fig. 3.8. Distribution of lynx observations in Austria from 1977 to 1983. After the release in Turrach (red star), the lynx distributed over a large range. Obviously, the animals therefore lost contact to each other so that there was no reproduction leading to the development of a population. Source: Breitenmoser & Breitenmoser-Würsten 2008.

Italy. A first attempt to bring the lynx back to the Italian Alps was made when two male lynx from the Carpathians were released in the Gran Paradiso National Park in the western Italian Alps in 1975. Both animals disappeared and one was later found dead in the French Alps (Boitani & Francisci 1978). According to Ragni et al. 1998 the first evidence of lynx immigrating to Italy from the reintroduced populations in the neighbouring countries dated back to 1980–1982. (The origin of the lynx in the Italian Alps was never confirmed, e.g. by means of genetics.) Since then, the area occupied by lynx has progressively increased: since 1982 in eastern Trentino-Lagorai; from 1985 in the Carnic-Julian Alps; from 1990 in western Trentino-Adamello Brenta and eastern Alto Adige; from 1992 in the Belluno Dolomites; from 1991 in Ossola Valley and Aosta Valley (these however being scattered observations, Molinari et al. 2001). Six reproductions were observed, however there were also two records of illegal killings and information about additional six cases of lynx poaching. In 1995, the size of the population (in the central-eastern Alps) was estimated at about 21 individuals (Ragni et al. 1998).

According to Molinari (1998), the first lynx observation in the Julian Alps close to the border with Slovenia was made as early as 1979, however there were hardly any signs until lynx came back at the beginning of the 1990s. The first lynx were believed to have immigrated to the northern part of Friuli-Venezia-Giulia from Austria. An increase and the distribution in the signs of presence showed also a south-westerly expansion (Fig. 3.9). The reports from the Julian Alps and Prealps also increased in that time. After 1992 lynx seemed to have been immigrating from Slovenia (Molinari 1998). In the north-east of the Friuli V.G. region (Tarvisiano) lynx was considered to have established a regular occurrence (Molinari 1998).

France. The lynx in the French Alps originated from the reintroductions in Switzerland, most likely not only from the Swiss Alps, also from the Jura Mountains through Gauges, Chartreuse or Salève (Stahl & Vandel 1998). From 1974 to 1994 there were 70 records of lynx of which the first ones came from the Chablais (Stahl & Vandel 1998; see also habitat models in Chapter 7.2). Regular records were reported from the Aravis Mountains. Over this period a southward expansion of about 200 km was observed, the increase in distribution became however only evident after 1989. The expansion did not lead to a continuous distribution area, but observations remained rare and widely scattered. Stahl & Vandel (1998) speculated that this unusual distribution was a consequence of insufficient monitoring efforts rather than lynx not being present.

Germany. A natural recolonisation of the German Alps by lynx was expected sooner or later from Switzerland or Austria (Kaczensky 1998), however up to now, this did not happen (see also Chapter 3.2.2).

3.2.2. Development from 1995 to 2009

Since 1995, the expert group “Status and Conservation of the Alpine Lynx Population” (SCALP, Chapter 4.1) has compiled and released data on the development of the Alpine lynx population and regularly published status reports (Fig. 3.9; Table 3.3):

1998 in the Italian journal Hystrix, covering the time of reintroduction or reappearance, respectively, until 1994 (Breitenmoser et al. 1998, Cop & Frkovic 1998, Huber & Kaczensky 1998, Kaczensky 1998, Molinari 1998, Ragni et al. 1998, Stahl & Vandel 1998) (Chapter 3.2.1);

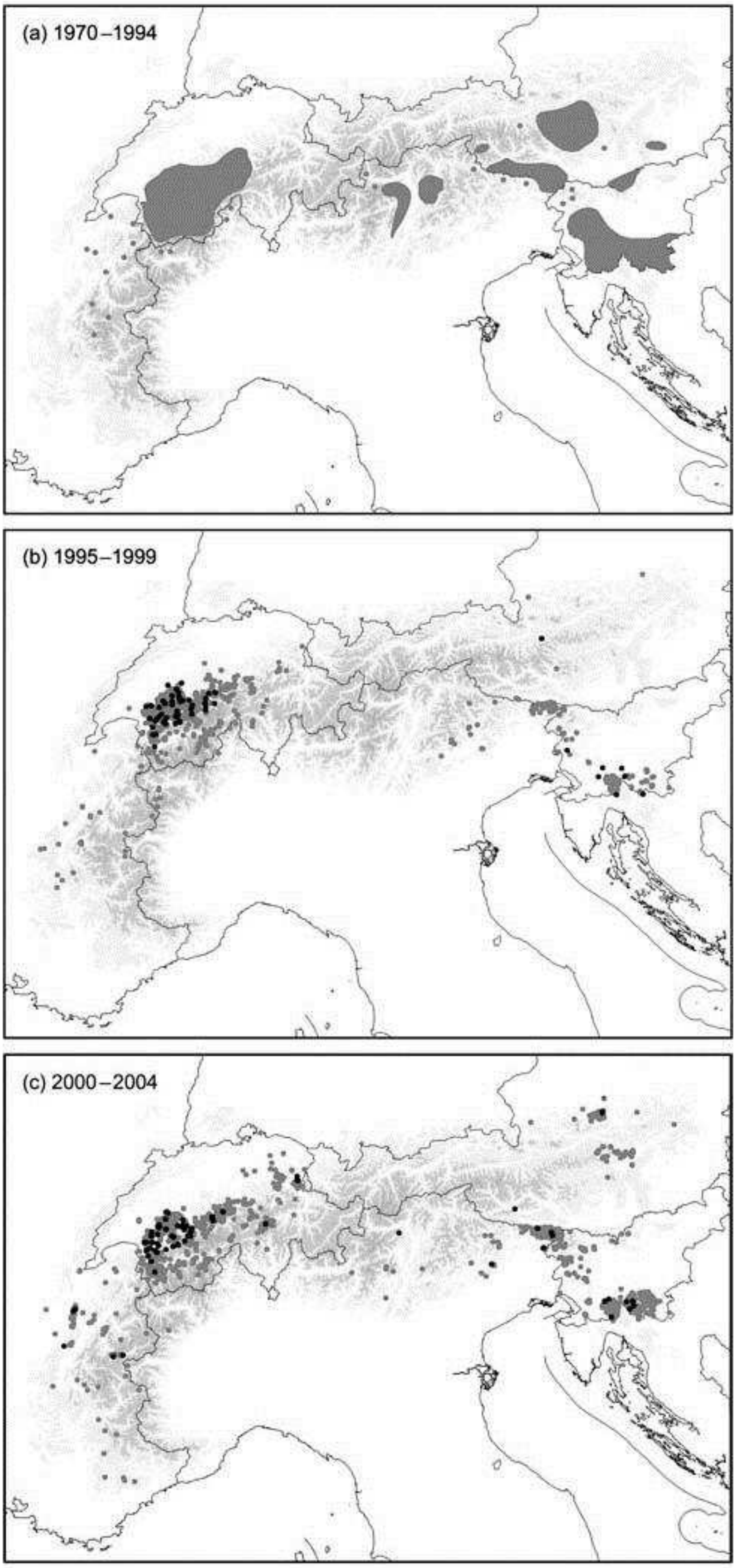


Fig. 3.9. Lynx distribution in the Alps in a) 1970–1994, b) 1995–1999, and c) 2000–2004. In b) and c) the black dots are Category 1 data, the grey dots Category 2 data (Chapter 4.1. for further explanation on the Categories). Source: Molinari-Jobin et al. 2010a.

2001 in the Italian journal Hystrix, covering the years 1995–1999 (Fasel 2001, Huber et al 2001, Molinari et al. 2001, Molinari-Jobin et al. 2001, Stahl & Vandel 2001, Stanisa et al. 2001, Wölfl & Kaczensky 2001);

2006 in the journal Acta Biologica Slovenica, covering the years 2000–2004 (Fasel 2006, Laass et al. 2006, Koren et al. 2006, Marboutin et al. 2006, Molinari et al. 2006, Molinari-Jobin et al. 2006, Wölfl 2006);

2011/2012 in the journal Acta Biologica Slovenica, covering the years 2005–2009 (Wölfl & Wölfl 2011, Zimmermann et al. 2011, Frick 2012, Fuxjäger et al. 2012, Kos et al. 2012, Marboutin et al. 2012, Molinari et al. 2012).

Table 3.3. Lynx population estimations 1995–2011 per country and the total for the Alps. Sources: Breitenmoser et al. 2000, von Arx et al. 2004, SCALP 2008, 2009, 2010, 2011, 2012.

Country	**Year(s)**						
	1995	**2001**	**2005-2007**	**2006-2008**	**2007-2009**	**2008-2010**	**2009-2011**
France	unknown	few ind.	15	15-20	15-20	15-20	15-20
Italy	~10	10-13	10-15	10-15	10-15	10-15	10-15
Switzerland	100	70	50-80	50-80	50-80	100-120	100-120
Liechtenstein	0	0	0-1	0-2	0-2	0-2	0-2
Germany	0	0	0	0	0	0-1	0-1
Austria	few ind.	20	5-10	8-14	7-12	6-12	6-12
Slovenia	n. a.	10	4-8	5-10	5-10	5-10	5-10
TOTAL Alps	**~120**	**~120**	**80-150**	**88-141**	**87-139**	**136-179**	**136-180**

The most relevant developments and events per country for the phase 1995 until 2009 were:

France. From 1995 to 1999, the number of records as well as the distribution area of lynx in France increased and covered 3,636 km² in 1999 compared to 1,215 km² five years earlier (Stahl & Vandel 2001). This increase was however assumed to be partly a consequence of improved monitoring (more people trained). Lynx presence was recorded in the major forested regions of the Prealps (foothills, e.g. Chablais, Glière/Aravis, Bauges, Chartreuse, Vercors, Diois/Beauchène), in the Chamonix and Maurienne valleys and the Briançon region. The distribution remained scattered and no large continuous area of presence existed. Whether this was due to an underestimation of lynx presence caused by difficulties in the detection of field signs or whether lynx was actually absent, remained unclear (Stahl & Vandel 2001).

During the period 2000–2004, numbers of lynx signs collected increased again. According to Marboutin et al. (2006) this was reflecting both a higher sampling rate and an actual north-to-south colonising process by the lynx. While north of Grenoble (Chartreuse/Epine massif, the Maurienne valley, and the Bauges massif) the range was now more or less continuously occupied and documented by quite robust data, southward only islets of presence mostly based on direct observations were detected. The southernmost observations were recorded in the Haut-Verdon close to the Mercantour Mountains. The corresponding population size was roughly estimated to be less than 20 individuals (Marboutin et al. 2006).

In the next pentad, different trends were noticed in the area with regular presence of lynx and in the area newly colonised: The area with regular presence increased until 2007 and then stabilised whereas the newly colonised areas which had increased until 2004, decreased from 2005-2009 (Marboutin et al. 2012). Robust lynx presence signs were still located mostly north to Grenoble. The southernmost reliable signs were collected in the Chartreuse and Vercors massif, and in the Maurienne valley. Overall, the French Alpine lynx sub-population was assessed to be stable and they estimated a regular population range of less than 1,350 km² and a population size of not more than 10–15 resident adults (Marboutin et al. 2012). The area newly colonised was assumed to be a mixture of actual dispersers and "phantom lynx" and therefore, a conservative approach was suggested by the authors, i.e. not considering such areas in the population status assessment as long as they do not turn to regular presence areas (Marboutin et al. 2012).

Italy. From 1995–1999 the positive trend observed earlier in the north-east of Friuli V.G. (Tarvisiano) (Chapter 3.2.1) halted and a decrease was noticed. At the same time an increased number of data was reported from north-east of the Veneto (Province of Belluno). Consequently, in the south-eastern Alps more data were collected over a bigger area than in the previous pentad (Molinari et al. 2001). The lynx occurrence of unknown origin in the Trentino (Chapter 3.2.1) went extinct again whereas the suspected presence of lynx in the Val d'Aosta and in the northernmost Piemonte (Val d'Ossola) was confirmed. With the exception of the new occurrence in the province of Belluno, lynx occurred only in areas bordering with Slovenia and Switzerland where populations exist (Molinari et al. 2001).

In the early 2000s lynx signs increased. This trend was however partly explained by improved monitoring effort (Molinari et al. 2006). Most of the presence signs (84%) were still concentrated in the eastern Italian Alps in Friuli V.G. and the province of Belluno. A few confirmed lynx signs indicated a colonisation of the Trentino-Alto Adige region. In the western Alps (Piemonte region), most signs of lynx presence were concentrated close to the French border. Molinari et al. (2006) roughly estimated the number of lynx occurring in Italy at this time to less than 20 individuals.

In the next pentad (2005–2009) the lynx signs decreased again and the area occupied by lynx diminished by one third (Molinari et al. 2012). The confirmed signs of lynx presence were confined to three concise areas: the north-eastern Alps of Friuli V.G., the Trentino province and the Ossola valley in the Piedmont region. No signs of reproduction were found. Some additional unverified signs were reported in the Belluno province, South Tyrol and in the western Alps close to the French border. Less than 15 individuals were estimated for the entire Italian Alps (Molinari et al. 2012).

Switzerland. From 1995 to 1999, more than 1600 signs of presence were recorded in the Swiss Alps. The number of lynx found dead more than doubled compared to the previous five years, indicating an increased population, particularly in the north-western Swiss Alps where most of the mortalities were recorded (Molinari-Jobin et al. 2001; see also Fig. 5.44 in Chapter 5.5.1). A moderate presence of the species was found in the central and south-western Alps of Switzerland whereas none or hardly any lynx signs were found in the eastern Alps. Based on a radio-telemetry study – between 1997 and 1999, 40 individual lynx were radio-collared – and further signs data available, Molinari-Jobin et al. (2001) estimated the total number of lynx at 70 individuals.

To counterpart the uneven distribution of lynx in Switzerland, 3 male and 3 female lynx were translocated from the north-western Alps to the eastern Swiss Alps (Ryser et al. 2004), as the expansion of the Swiss lynx population is crucial for the conservation of the lynx in the whole Alps (Molinari-Jobin et al. 2001).

The removal of individuals for this translocation project together with the removal of a stock-raider plus several cases of illegal killings clearly reduced the lynx population in the north-western Alps in the pentad from 2000–2004 (Molinari-Jobin et al. 2006). Nevertheless, this compartment remained the area with the highest lynx density within Switzerland. In the Valais and central Switzerland the trend was slightly positive. Due to the translocation project, the distribution of lynx in the Swiss Alps had considerably increased. All together, the population of lynx in Switzerland was considered as stable with 60–90 individuals estimated, of which more than 50% occurred in the north-western Swiss Alps (Molinari-Jobin et al. 2006).

In 2003, the newly established sub-population in the north-eastern Swiss Alps was restocked by another 3 lynx from the Jura population (Ryser et al. 2004). The success of the project was yet believed to be doubtful as several losses had been reported and the number of lynx was only estimated at 4–5 individuals (Molinari-Jobin et al. 2006). In the winters 2006/07 and 2007/08 the translocation of another three lynx to north-eastern Switzerland was therefore carried out (A. Ryser, pers. comm.).

There were still important areas with very few lynx records only (e.g. the Grisons and Ticino). The confirmed records might have originated from single individuals who left the core population. Such individuals can produce signs of presence at low density and over huge areas, as they search for an area to settle down (Molinari-Jobin et al. 2006).

Even though the number of lynx presence signs remained almost stable from 2005–2009 compared to the previous pentad, there was a 7.6% increase in the area occupied (12,637 km²; Zimmermann et al. 2011). The north-western Swiss Alps remained the region with the highest number of chance observations. It was followed by the compartments central Switzerland west and north-eastern Switzerland. These sub-populations acted as source in the current pentad, as signs of reproduction were reported almost every year. The translocation to north-eastern Switzerland was still the only significant contribution to the spatial increase of the lynx range in the last 10 years in the Swiss Alps (Zimmermann et al. 2011) as well as in the whole Alps, respectively (Molinari-Jobin et al. 2010a). The status of the sub-population in the Valais was less clear. As only few signs of reproduction and mortalities were reported over the pentad, it seemed indeed to be a sink population. From the few signs of lynx presence reported in the remaining compartments (Grisons, central Switzerland east and Ticino) the authors concluded that only a few single lynx that did not yet establish the typical social organisation occurred there (Zimmermann et al. 2011). An occupancy-based population estimate from a parallel study resulted in about 111 (SE = 10) independent lynx for 2005–2009.

Liechtenstein. The area of Liechtenstein (160 km²) will only allow to host from 1 to exceptionally 3 adult lynx (Fasel 2001). The first observations of lynx in Liechtenstein occurred in January 2004 and January 2005 (Fasel 2006). Before, neither direct observations, nor livestock killed, tracks or other signs of presence had been recorded since the historic extinction of the species (Fasel 2001). The locality of the observations, the forest of the village Schaanwald near the border to Austria was very near to observations in Vorarlberg (Laass et al. 2006) and it was therefore assumed potentially to be the same animal (Fasel 2006). As in neighbouring Switzerland, translocations to the north-eastern Swiss Alps had started in 2001/02 (Ryser et al. 2004, see above), it was assumed that these observations relate to dispersing lynx from the newly created population. Some lynx signs (two tracks and one sighting) were recorded in Liechtenstein between 2007 and 2009 (Frick 2012). In March 2013, female lynx HEIA which was equipped with a GPS-collar visited Liechtenstein, coming from the canton of Grisons. She lost her collar in April 2013 back in the Grisons (A. Ryser, pers. comm.).

Germany. Besides a few unconfirmed direct sightings and rumours there was no confirmed evidence of lynx presence in the German Alps (Wölfl & Kaczensky 2001, Wölfl 2006, Wölfl & Wölfl 2012). The nearest lynx sub-populations to the Germans Alps are found in north-eastern Switzerland (distance 70 km) and in Slovenia (distance 180 km), besides the population in the Bavarian-Bohemian Forest, which is however separated from the Alps by open agricultural land. While single individuals might have visited the area (Wölfl & Wölfl 2012), a natural colonisation within the next decades followed by a successful settlement of a population seems very unlikely (Molinari-Jobin et al. 2010b).

Austria. From 1995 to 1999 lynx records were widely distributed almost all over the Austrian Alps with the exception of Vorarlberg and Tyrol in the west. The Hohe Tauern, Gailtaleralpen and Nockberge in the north-western part of Carinthia, as well as the Karnische Alpen along the Carinthian/Italian border seemed to be the centres of lynx activity. A remarkable increase of reported lynx observations and other lynx signs was observed in Upper Austria, particularly the Upper Austrian Kalkalpen. However, across Austria there was only one "hard fact" (Category 1 observation) – a male lynx killed in a traffic accident in southern Salzburg in 1995 – and no confirmed reproduction was recorded. The data did not base on a systematic monitoring, and there was clearly no established lynx population but rather some scattered individuals at best (Huber et al. 2001).

In the next pentad (2000–2004) the number of lynx signs collected almost doubled. However, the area of distribution of the records shrunk and the confirmed records originated mainly from two distinct areas: the National Park Kalkalpen in Upper Austria and the Niedere Tauern mountain range in Styria (Laass et al. 2006). Again, there was no proof of lynx reproduction in the Austrian Alps. Unconfirmed records were distributed over a greater area and included data from the north-eastern Limestone Alps, north-western Carinthia and Vorarlberg (the latter may have been dispersing individuals from the reintroduction in eastern Switzerland). Due to missing good data on lynx over large areas of the potential range in the Austrian Alps, Laass et al. (2006) felt unable to evaluate the actual distribution and the population status.

In the subsequent five years, the spatial distribution and the number of records collected remained stable. The distribution of the signs showed three clusters: (1) the clearest in Upper Austria (Kalkalpen National Park) with 85% of confirmed records, (2) in Styria (Niedere Tauern), and (3) in southern Carinthia (Karnische Alpen). With the exception of the National Park Kalkalpen the collection and confirmation of lynx signs of presence in the Austrian Alps depended however on private initiatives of a small number of interested individuals. From other regions than the three mentioned, only isolated or unverified records were reported (Fuxjäger et al. 2012). The scattered observations still indicated the presence of single individuals only. Fuxjäger et al. (2012) estimated 5–10 individuals to be present in the Austrian Alps from 2005–2009.

Slovenia. Stanisa et al. (2001) defined 4 regions of lynx presence in Slovenia: (1) the southern part of the country comprising the area south-east of the Ljubljana-Trieste highway (Kocevska, Notranjska) where lynx were reintroduced (Chapter 3.2.1) and the lynx number was still the highest; (2) the western part of the country with the Julian Alps, where lynx had started to immigrate in the mid-1980s; (3) the Karavanke and Kamnisko-Savinjske Alps in the north of the country, as well as some other isolated areas where only a very low number of lynx signs of presence were collected, supposedly from single individuals; (4) the fourth region comprised the north-east of Slovenia where lynx was still absent. From 1995–1999, the area of lynx occurrence did not increase. The annual hunting quota (though considerably reduced in these years) had not been reached since 1992, indicated that the lynx population was dwindling. The core of the population in southern Slovenia had shifted slightly to the west. The number of lynx in southern Slovenia – belonging to the Dinaric lynx population (von Arx et al. 2004, Linnell et al. 2008) – was estimated 30–40 individuals, while in the west of the Jesenice-Ljubljana-Trieste highway (part of the Alpine population) about 10 individuals were estimated to be present (Stanisa et al. 2001). The Ljubljana-Trieste highway was considered the only potential habitat barrier separating these two population segments. Stanisa et al. (2001) assumed lynx would cross the highway as brown bears regularly do. The data available did however not allow evaluating the magnitude of lynx dispersal into the Alps.

During the next pentad (2000–2004) the population stabilised (Koren et al. 2006). The recorded signs of lynx presence increased (due to an improved monitoring), the range however remained more or less the same. Hunting mortality was decreasing, approaching zero. In the north-western (Alpine) subpopulation, two areas with more abundant lynx signs were identified, Tolmin and Bovec, respectively. In these areas, increasing attacks of livestock by lynx confirmed the presence of the predator. This subpopulation stretched into Italy, and to the triangle Austria, Slovenia and Italy, and to Kepa and Mojstrana at the border with Austria. The area of lynx presence in Kamnik–Savinja Alps was smaller, but had also increased slightly. A total of 30–50 lynx were estimated for Slovenia, of which 15 in the Alpine part (Koren et al. 2006). At this time, Koren et al. (2006) judged the potential for an expansion of the population across the south-eastern Alps positively.

However, only a few years later Potočnik et al. (2009) believed that an expansion toward north and northwest into the Slovenian Alps had probably slowed down due to significant spatial obstacles (traffic infrastructure, urban areas, open habitat) separating the Alps from the Dinaric region. In addition, no reproduction had ever been recorded in the Slovenian Alps up to then (Potočnik et al. 2009). From 2005–2009, both the distribution of lynx as well as the number of signs remained similar (Kos et al. 2012). Actually, there was no major change in the range occupied by lynx in the past 15 years. However, there was a decrease in the relative population density during the last years of the current pentad, particularly in south-eastern Slovenia (Kočevsko region, which was still considered the source population) and in the Alps. Kos et al. (2012) estimated only 15 to 25 resident lynx remaining in Slovenia. They assumed this lynx population to be Critically Endangered because of demographic and genetic problems (Chapter 4.2.2). Complete legal protection of lynx was adopted in 2004 (Kos & Potočnik 2013). To prevent local extinction, according to Kos et al. (2012) an active approach addressing demographic factors as well as improving the depleted gene pool is needed for a revitalisation of the population.

3.3. Recolonisation of the Alps by the wolf and population development

3.3.1. Recolonisation of the western Alps from the Abruzzo population

Zimen & Boitani (1975) estimated that in 1973 only 100 wolves were left in the Italian Apennines and they considered the species to be highly endangered. The northern limits of distribution were the Sibillini Mountains in the central Apennines and the population was divided into small isolated groups (Fig. 3.5). A ministerial decree had given the wolf full protection in 1971 (Zimen & Boitani 1975). This, together with an increase in prey base (partly through reintroductions, Chapter 5.3) helped the wolf to increase its range (Boitani 1992). By 1990 they had already reached Liguria in north-western Italy and Sila (Calabria) in the very south of the country (Francisci et al. 1991). Genetic data from samples from the Maritime Alps and the Ligurian Alps indicated that wolves colonising the south-western Alps originated exclusively from the Italian source population (Lucchini et al. 2002.).

It was in November 1992 when for the first time wolves were confirmed in France, in the Mercantour National Park, Maritime Alps (Houard & Lequette 1993). However, there were earlier observations of canid-like animals in the region (Poulle et al. 1999). During winter 1992/1993 it became evident that a wolf pair had settled, which reproduced in the following season (Fig. 3.10; Poulle et al 1999). From 1994 to 1995, 129 wolf attacks were recorded in the Mercantour (Dahier & Lequette 1997). By 1999, there were already four packs installed in the park and the number had increased from two wolves in 1992 to 19 in 1999 (Fig. 3.10; Poulle et al. 1999, Poulle et al. 2000). By that time wolf presence had been confirmed also outside the Mercantour National Park in the Queyras (Hautes-Alpes), Savoie, Isère, Drôme, Var and Alpes-de-Haute-Provence (Poulle et al. 1999).

In 1995 and 1996 numerous attacks on sheep occurred in the area of the Great Saint Bernhard, Valais, Switzerland. Investigations in the field revealed wolf to be responsible for the kills (Landry 1997a). Genetic analysis of scats found nearby confirmed the presence of two different wolves of Italian origin (Taberlet et al. 1996; see Chapter 4.3.2 for information about the genetic differentiation of wolf populations). One of the individuals had earlier been identified in France in the Mercantour National Park and in the region of Grenoble. These results confirmed the continued spread of the wolves of Italian origin towards the north (Taberlet et al. 1996). At least three wolves stayed in the Canton of Valais (Switzerland) between 1998 and 2000 (Crettenand & Weber 2000).

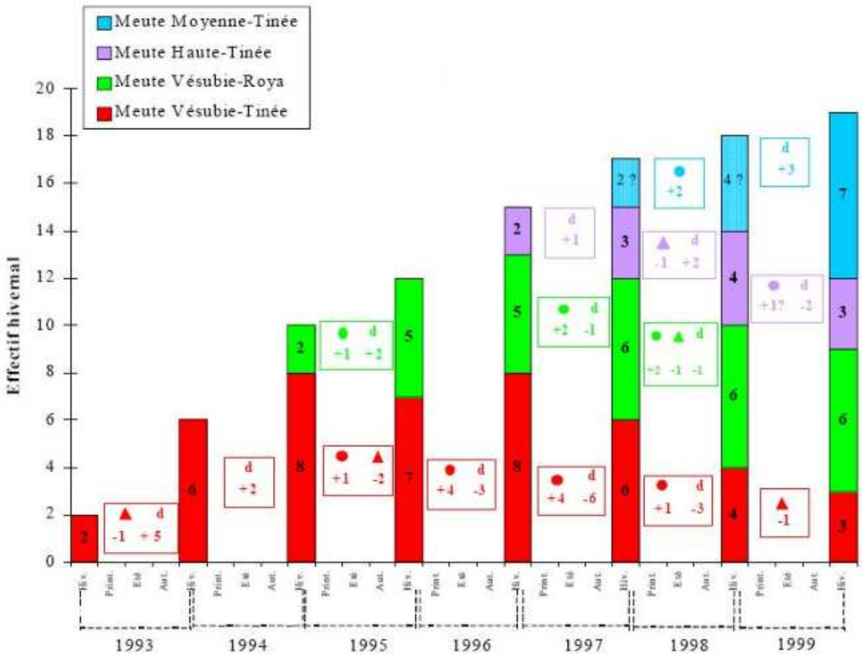


Fig. 3.10. Evolution of the winter numbers of wolf individuals in the packs of Mercantour National Park from 1993 to 1999. Numbers were estimated through snow-tracking and direct observations. Circles = number of offspring observed, triangles = number of wolves found dead, (d) concerns the minimum number of individuals which are added (+) or subtracted (-) to the pack in the course of the year (?: uncertain data) (Poulle et al. 2000).

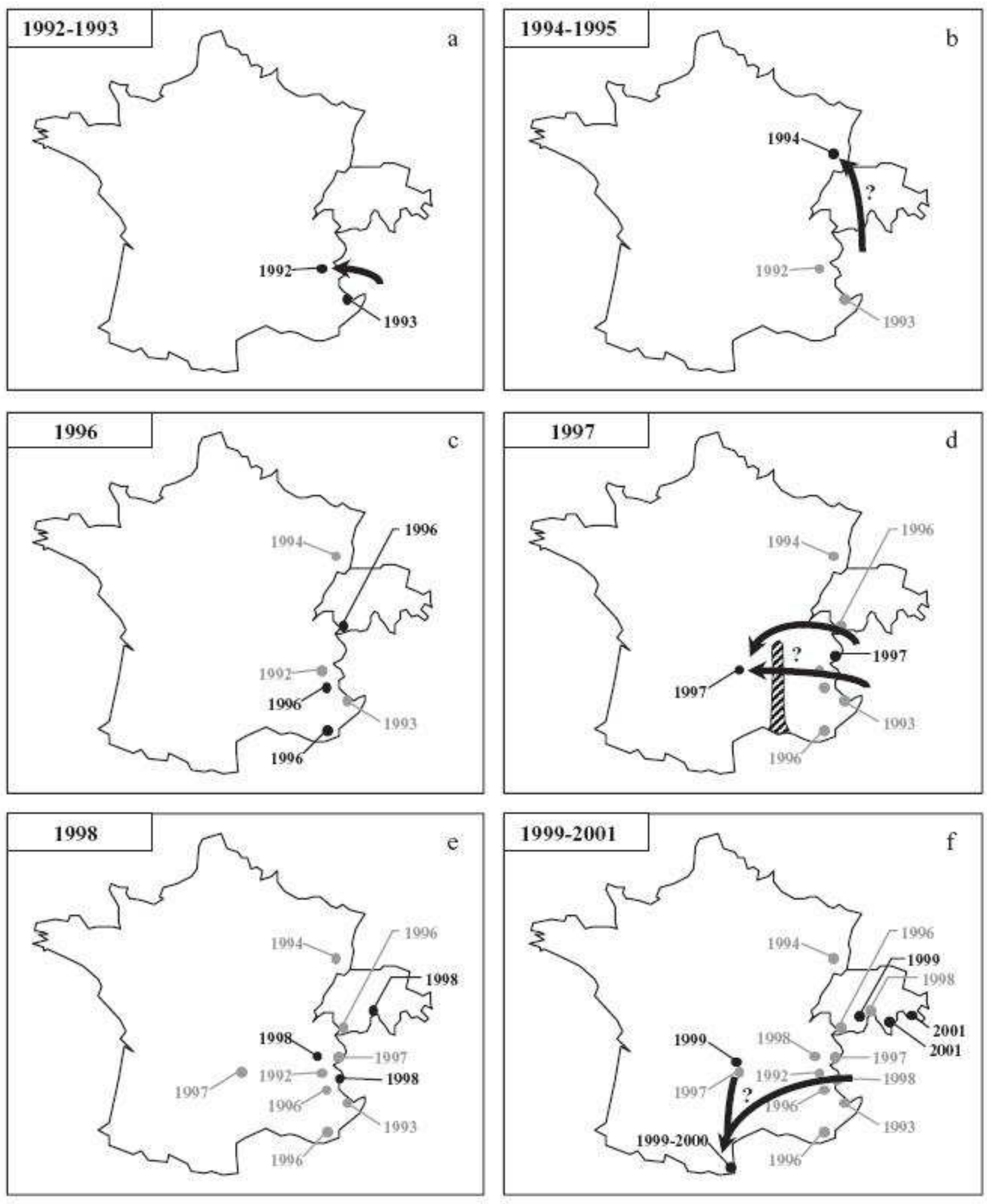


Fig. 3.11. Time of first genetic detection (black dots) of wolves of the Italian lineage in different regions for France and Switzerland. Arrows correspond to putative colonisation routes. Cross-hatched areas indicate regions of high human activity that might act as a barrier to dispersal (Valière et al. 2003).

Valière et al. (2003) analysed assumed wolf samples from France and Switzerland collected from 1992 to 2001 genetically. Besides demonstrating the applicability of the method for species identification and assignment to the population of origin, the work also confirmed the recolonisation pattern in the two countries by the time of first genetic detection in a region (Fig. 3.11). The presence of (male) wolves from the Italian lineage in locations far away from established packs seemed to be the rule rather than the exception (Valière et al. 2003) due to the strong dispersal capacity of the species (Chapter 7.1).

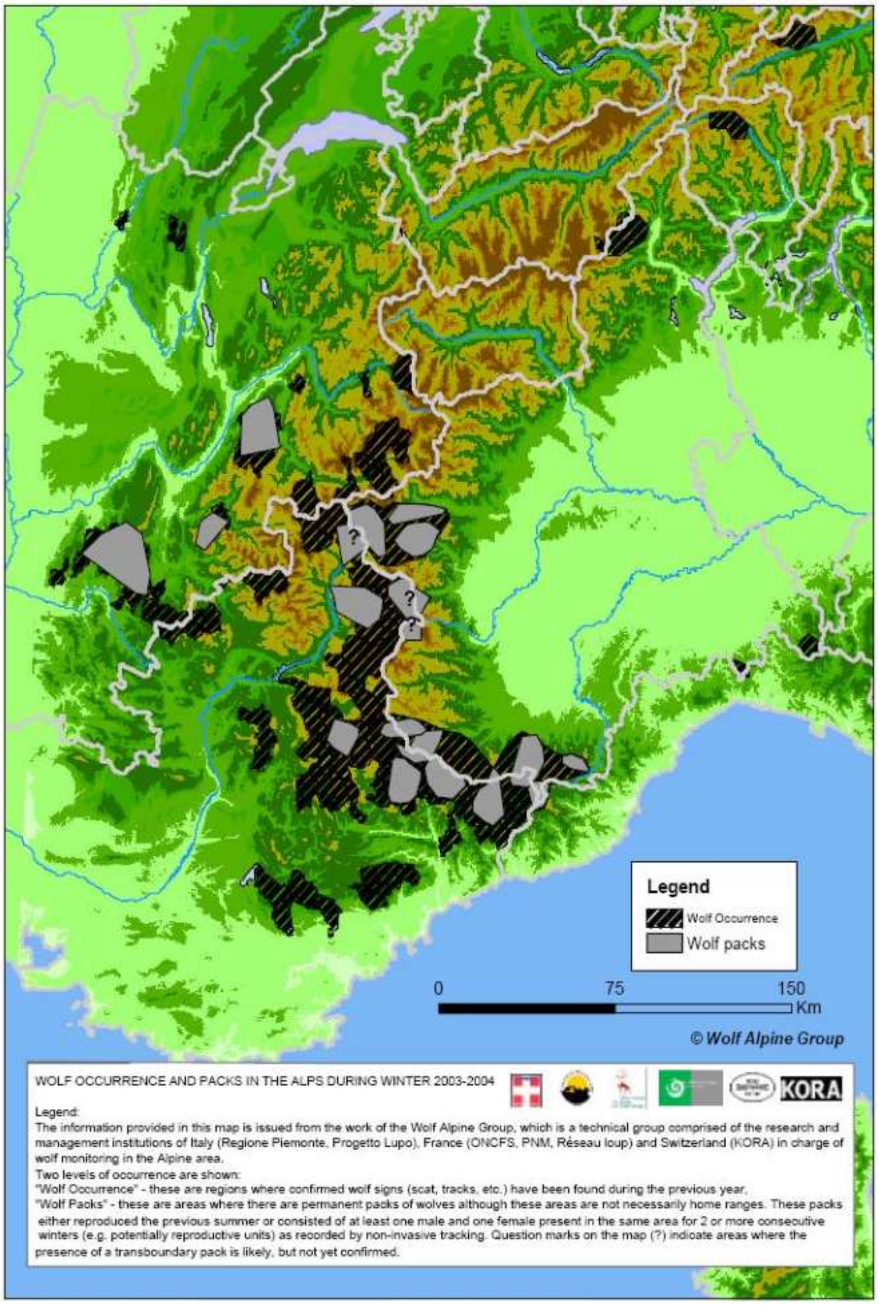


Fig. 3.12. Wolf occurrence and packs in the Alps during winter 2003-2004 (WAG in Marucco 2005).

While a population estimation for the Alps was not yet possible for 1997 (numbers for Italy were only available in total and not per population; Boitani 2000), in 2005 the population in the south-western Alps was estimated to be around 100–120 individuals (Linnell et al. 2008). The distribution at that time is shown in Figure 3.12.

3.3.2. Recolonisation from other source populations

Besides the Apennine population, other wolf populations in Europe are situated within possible wolf dispersal distance from the Alps. Indeed, the Alps could be reached from wolves from the Central-European Lowlands population (with eastern Germany as closest range region), the Carpathian population (Slovakia and Czech Republic) and the Dinaric-Balkan population (Slovenia) (Fig. 3.13).

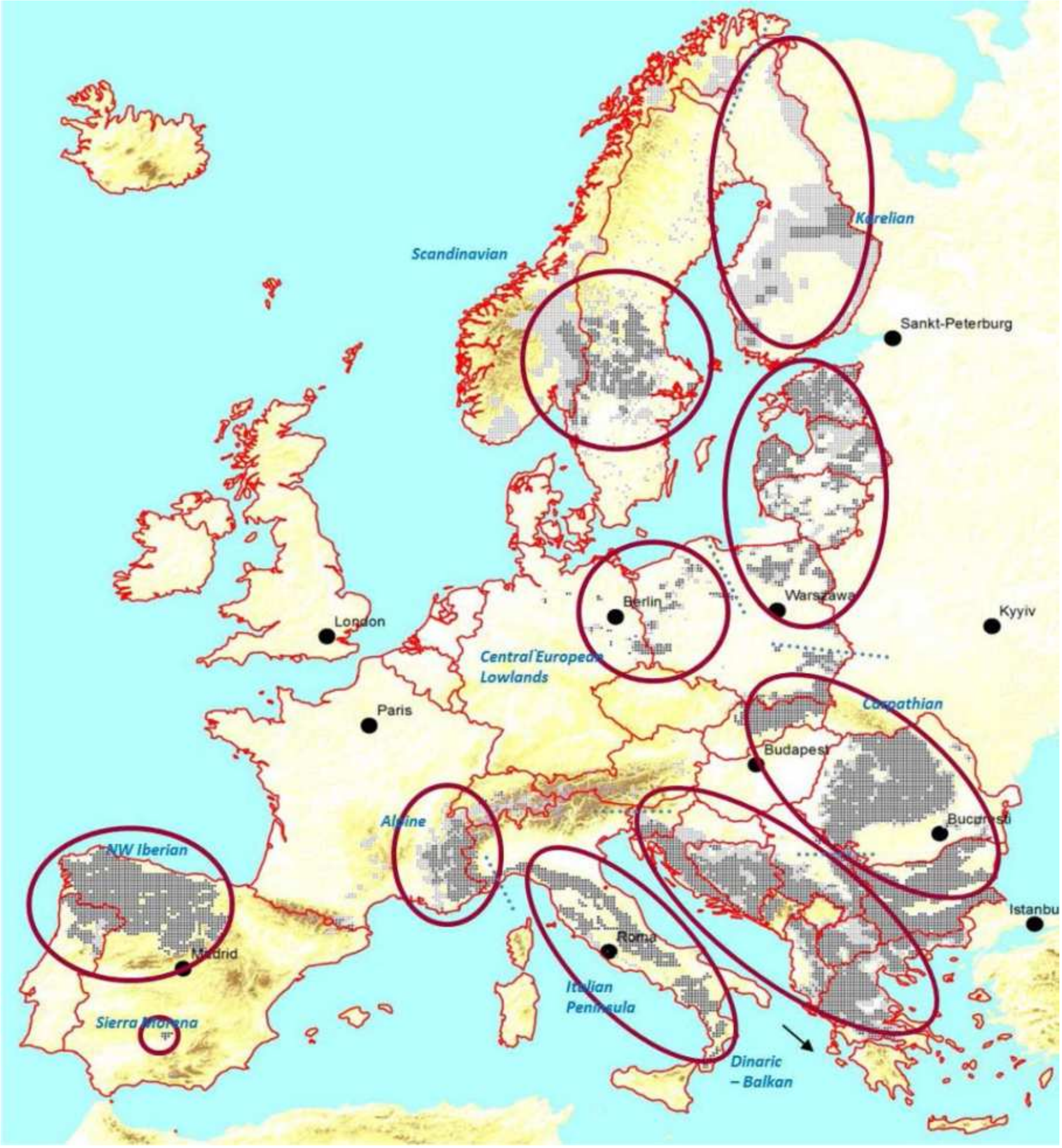


Fig. 3.13. Wolf distribution in Europe 2011. Dark grey cells: permanent occurrence, light grey cells: sporadic occurrence. Red borders mark countries for which information was available. Circled are the populations as defined by the IUCN/SSC Large Carnivore Initiative for Europe. Source: Kaczensky et al. 2013a.

Austria lies in the centre of these populations (Fig. 3.13). According to Zedrosser (1996) there are three major migration routes into Austria for wolves: (1) Slovenia-Carinthia-Styria, (2) Czech Republic–Mühlviertel and (3) Slovenia–Italy–Carinthia, which is considered the most important route for the resettlement of Austria by wolves (and bears) (Zedrosser 1996). Until 1973 there were regular immigrations of wolves into Styria and an occasional further dispersal from there towards Carinthia and eastern Tyrol (Dungler 2006). A single male wolf was killed illegally in January 1996 in Upper Austria. This was by then the first wolf after more than 20 years (Zedrosser 1996). After several sightings of a wolf in 2005, a photo was taken in June the same year in the border region Styria–Carinthia (Dungler 2006).

The number of single dispersing wolves showing up in Austria has increased since. From 2009–2012 at least 11 individuals were detected genetically. These wolves were found almost all over the country independent of the population they were originating from. Wolves from the Western Alps were found from Tyrol to Lower Austria, wolves from the Carpathian and the Dinaric Mountains reached areas far north and west in Austria. Austria may therefore indeed develop into a cross-breeding area of wolves from different distinct populations (Rauer et al. 2013; see also Fig. 4.17 in Chapter 4.3.1).

In 2013, the first wolf pair of mixed population origin reproduced in the Lessinia region, Italy (Parco Naturale Regionale della Lessinia 2013). The male "Slavc" (GPS collared by the SloWolf project; see www.volkovi.si/?lang=en) had immigrated from Slovenia (Dinaric-Balkan wolf population) and met with a female wolf of the Italian lineage. They are still in the region and had again offspring in 2014 (Regione Veneto 2014).

Besides receiving wolves from different populations, animals from the Alpine population are also emigrating to other areas. Since 2002, several wolves have spread from the French Alps to the Pyrenees, to the Jura Mountains, and since 2012 to the Vosges Mountains and the Palatinian Forest in Germany, recolonising new areas (Marboutin 2013a).

3.4. Discussion and conclusions

For the Alpine lynx population, the most important area is in the north-western Alps (western Switzerland), followed by north-eastern Switzerland and the south-eastern Alps (Italy and Slovenia). Both populations are the result of reintroductions in the early 1970s with very few founder animals, and both populations show today a high inbreeding coefficient (Chapter 4.2.2). Two other smaller nuclei lie in the Chartreuse (France) and the Kalkalpen region (Upper Austria). However, only in four areas reproduction was reported: north-western Swiss Alps, Friuli, north-eastern Switzerland and the Chartreuse region (Molinari-Jobin et al. 2010a, b). More recently, reproduction also occurred in the Kalkalpen (Fuxjäger 2014). There is no permanent lynx presence with reproduction between or outside these areas, and even single confirmed observations are rare[3] (Molinari-Jobin et al. 2010a, b).

The wolf made a remarkable come-back to the Alps. Within only two decades the species settled in the French Alps and the western Italian Alps from the Apennine population and started to recolonise the Swiss Alps. But wolf populations are increasing across Europe and some of these populations are also expanding, so that all regions of continental Europe are again within the reach of dispersing wolves. The eastern Alps lie in the centre of several large wolf populations and could act as a melting pot of European wolves in the near future. Thanks to non-invasive genetic monitoring, this process can be shown – given the data are processed equally between the different countries. The recolonisation of the Alps by the wolf is requiring cross-border cooperation and the regular exchange of monitoring data. Since 2001, the Wolf Alpine Group (WAG; Chapter 4.1.2) holds regular workshops (reports of the WAG workshops are available under www.kora.ch).

[3] Hunters have reported reproduction from Trnovski gozd, Slovenia, in 2013 (M. Jonozovic, pers. comm.). However, these remain unconfirmed (category C3, Chapter 4.1.1.). It seems that there is no systematic monitoring established in this area as, in fact, all reports of lynx occurrence from Trnovski gozd were from the C3 category and as such unconfirmed (Kos et al. 2012).

4. Present situation and assessment of the Alpine wolf and lynx populations

4.1. Monitoring of lynx and wolf across the Alps

Hellawell (1990) proposes the following definition for monitoring: "A regular and structured surveillance in order to ascertain the compliance of a measure with an expected norm or standard (goal) to be reached (e.g. recovery of an endangered population to a viable status)." In the context of wildlife management, “monitoring” often refers to the continuous surveillance of a population without trying to meet a specific goal, maybe at best to inform an adaptive management process. The main aims of monitoring are to determine the *distribution* and *abundance* of the species studied, its *population trend*, *health*, and *genetic status*. Changes in the distribution of a species can be indicative for its conservation status (Marboutin et al. 2011). The monitoring can be adapted for short and long term projects. The SCALP categories (see below) were used to standardise the interpretation of lynx data compiled, and the SCALP system was adapted also for other species, e.g. for wolves in the Alps. As it is impossible to count every individual in a wildlife population, population sizes are extrapolated from the data available. Such an extrapolation process can be very different between species and regions, reaching from simple guesses to standardised and robust statistical analyses allowing also estimating the reliability of the estimation (e.g. by providing a standard error). Detection (leave alone “counting”) of large carnivores is difficult. Animals actually present can be missed because they are elusive (false negative), or signs and observations can be misinterpreted leading to believe that the species is present if indeed it is not (false positive). Indeed, both errors can influence the results of lynx and wolf monitoring, especially at an early phase of the recolonisation, when a certain area is not yet permanently settled and when the experience of local wildlife management institutions is still limited. Molinari-Jobin et al. (2012) have explored this phenomenon and its consequences for the lynx in the Alps. The only way to meet this challenge is to continuously improve the quality of the monitoring and to apply wherever feasible scientific robust methods to monitor wildlife populations.

4.1.1. Status and Conservation of the Alpine Lynx Population (SCALP)

Monitoring carnivores in landscapes such as the Alps is difficult; different countries use different methods depending on the population status and the local/national wildlife management organisations. The presence of wolf and lynx in the different Alpine countries were determined based on signs such as scats, prey remains, livestock predation, snow tracking, howling (for wolves), camera trapping (mainly for lynx), direct observations, which are collected by independent observers as well as during deterministic surveys. The amounts of data obtained vary across the countries as the survey efforts and numbers of trained observers differ. It is hence important that these diverse data sets are interpreted according to common rules and understanding.

For the lynx, this assignment is taken care of by the expert group of the SCALP (Status and Conservation of the Alpine Lynx Population) project (KORA 2014). The principle goal of the SCALP group is to support the conservation of lynx in the Alps through consistent monitoring and informing the relevant bodies. The monitoring considers a set of common rules and principles, taking into account opportunities and constraints of each country. Observations are classed according to the SCALP categories (Molinari-Jobin et al. 2003):

- Category 1: represents the "hard facts", e.g. all reports of lynx killed or found dead, photographs of lynx as well as young orphaned lynx caught in the wild and put into captivity;
- Category 2: incorporates all records of livestock killed, wild prey remains, tracks and scats reported by people who attended special courses. These records are mostly an objective proof of lynx presence;
- Category 3: includes all wild prey remains, scats and tracks reported by the general public as well as all sightings and vocalisations, e.g. signs that cannot be verified.

These categories can be applied retrospectively, allowing integrating each record into the monitoring data set and to classify an observation according to its expressiveness and reliability. The comparison of the distribution of records classified into the three categories allows assessing the performance of the monitoring system and closing gaps if detected.

4.1.2. Wolf Alpine Group WAG

In 2001, the Wolf Alpine Group WAG was created as a result of a workshop organised in France and which was attended by experts from Switzerland, Italy and France (WAG 2014). Today, the WAG is an informal group composed of experts from each of the Alpine countries. The aim of this scientific group is to exchange information, share methodologies, standardise monitoring, data collection and genetic approaches. They also take into account differences in data collection methods applied in different countries when interpreting the results from the national monitoring projects. New progress in technology and improved monitoring methods are also considered and implemented where possible.

4.1.3. Monitoring of lynx

France. A network of around 1200 trained field experts is in charge of presence signs surveys throughout the year (see also Chapter 4.1.4); any sign detected is recorded and sent to a central state agency, the *Office national de la chasse et de la faune sauvage* (ONCFS), which is in charge of the validation process (Marboutin 2013b). Since 2010, Capture-Mark-Recapture (CMR)-based estimates of abundance and density have been derived from four large study areas (several hundred km² each), which are intensively surveyed using camera-traps to identify individuals based on their coat patterns.

Italy. Signs of lynx presence are collected by a network of people, mainly game wardens and foresters who have attended special training courses. The number of trained people however varies regionally (Molinari et al. 2012). Whenever possible, these "lynx experts" verify the signs reported by the general public. One or two persons per region are responsible for the centralisation of the data and transfer of the data to a common database at the end of the year. The data are then verified according to the SCALP criteria. In the Alps of north-eastern Italy, camera-traps are used to identify individual lynx (Molinari et al. 2012).

Switzerland. Switzerland uses a stratified approach to monitor the lynx population. There is a stratification in space (national level, compartments and smaller reference areas within compartments), time (e.g. chance observations are gathered year round whereas systematic camera-trapping which is very labour intensive is conducted every 2 to 3 years in smaller reference areas) and in the datasets according to the type of observation and the SCALP category. The following data sources are available: (1) game warden observations for each canton, compiled once a year by means of a simple questionnaire (Fig. 4.1), (2) chance observations such as sightings and signs, known losses of lynx, number of compensations granted in cases of livestock killed by lynx, (3) opportunistic and (4) deterministic camera-trapping.

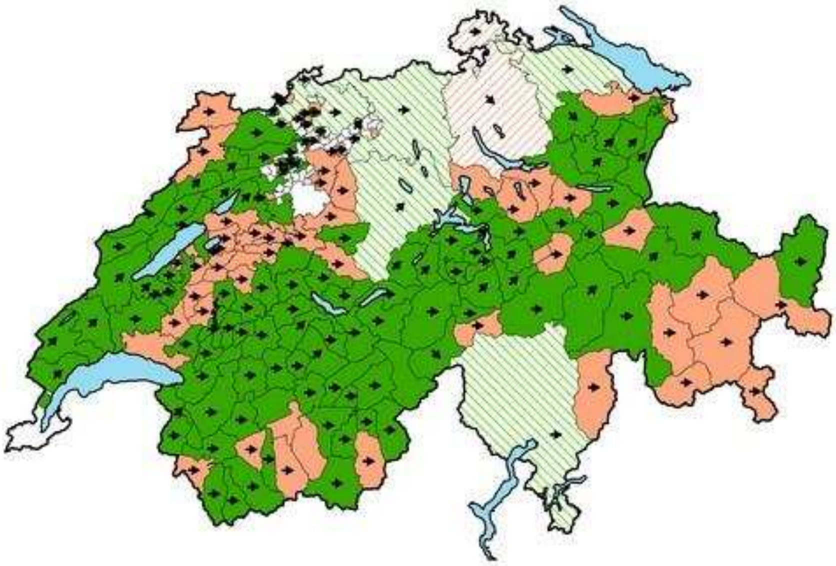


Fig. 4.1. Results of questionnaires filled in by game wardens in 2012. The green areas illustrate regions with lynx evidence, the red areas are regions without evidence. The arrows indicate the direction of evolution of the lynx population for the delimited area. The cantons with the striped pattern only have results for the entire canton while the solid colours show population status at a local scale and areas where no data was provided are marked in white (KORA 2014).

Radio-telemetry is used to address more specific research questions. Genetic samples of captured or dead lynx are collected and analysed. Each dead lynx is examined at the Veterinary Faculty of the University of Bern. The population size is estimated using data from photographic capture-recapture analyses (per reference area) and chance observations collected in occupied cells (von Arx & Zimmermann 2013).

Germany. Since 2009, experienced persons from the federal states meet once a year for a country-wide assessment of large carnivore population and distribution. The meetings are organized by the Federal Agency for Nature Conservation (Bundesamt für Naturschutz, BfN) and serve as the basis for the FFH reporting (Reinhardt et al. 2013). In 2011, the first national distribution map (10 x 10 km EEA grid) was produced showing lynx distribution in 2010, based on the SCALP C1 and C2 records. In Germany, all lynx signs reported by third parties are inspected and documented by a network of trained lynx inspectors (Kaczensky et al. 2013b). The majority of these inspectors are volunteers. These documented records are evaluated by 1–2 “experienced persons” (a wildlife professional with documented large carnivore experience) for each federal state and rated according to the SCALP criteria (Kaczensky et al. 2009). There is no lynx occurrence in the German Alps at present (Chapter 4.2.1).

Austria. Monitoring in Austria is done by collecting and assessing chance observations and data from opportunistic camera trapping (FaGrÖ 2009). Lynx signs from the Alps are compiled in a national database at the Kalkalpen National Park and new data are used to regularly update the SCALP Monitoring Report (SCALP 2012). Since 2011, the monitoring standards, which were adapted for Germany, have been adopted and large carnivore experts in Austria meet once a year to discuss the quality and distribution of lynx signs in Austria (Kaczensky et al. 2013c).

Slovenia. Monitoring is carried out by the Slovenia Forest Service using the SCALP methodology in an opportunistic manner in the State Hunting Grounds with Special Purpose. All dead animals are examined at the Veterinary Faculty of the University of Ljubljbana (M. Jonozovic, pers. comm.). Samples for genetic analysis are also collected and analyzed at the University of Ljubljana in order to further investigate population inbreeding (Kos & Potočnik 2013).

4.1.4. Monitoring of wolf

Similar monitoring techniques are applied across the Alpine countries. These consist of collection of chance observations and sign surveys, snow-tracking sessions in winter, wolf howling sessions in summer, which are all carried out in parallel with standardised non-invasive molecular tracking (WAG 2011), based mainly on wolf excrements and saliva gained from kills. A more intensive and deterministic monitoring protocol is applied in the south-western Alps of France and Italy, where a permanent wolf population with packs is present.

France. A network of around 1200 trained field experts carries out presence signs surveys throughout the year. All signs are described and sent to the ONCFS, which is in charge of the validation process. CMR-based estimates of abundance are derived from non-invasive sampling. Intensive snow tracking is implemented over every packs' territory. Wolf howling is implemented, when and where necessary, as a complementary field action to snow tracking, so as to get a more accurate estimate of the number of packs, and to update this number twice a year (end of summer & end of winter; Duchamp et al. 2012, Marboutin 2013a).

Italy. A combination of snow-tracking, non-invasive genetic sampling and wolf-howling techniques are used. Staff from the Forestry Service and rangers of the regional and national protected areas contributed to the monitoring and assessment of the population size and distribution of wolves over the years in the Piedmont region (Boitani & Marucco 2013).

Switzerland. The monitoring of wolf in Switzerland is carried out opportunistically. Samples for genetic analysis, livestock and wild prey killed, sightings and pictures are collected continuously. Genetic analysis of scats, saliva, tissue or hair allows for the identification of the species. When the sample is of good quality, individual fingerprinting can also be carried out (von Arx & Manz 2013).

Germany. Germany is a federal country and wildlife monitoring falls under the jurisdiction of the states (Länder). Since 2009, the Länder have adopted country-wide monitoring standards for large carnivores (Kaczensky et al. 2009). They specify how population size and trend, area of occurrence and distribution trend, and range of large carnivores are to be documented. The standardised protocol aims to create a uniform interpretation of the data collected. They define what signs of large carnivores are to be collected and under what conditions they qualify as hard evidence (C1), confirmed observation (C2), or unconfirmed observation (C3). To estimate population size and determine the area of occurrence only hard evidence (C1) and confirmed observations (C2) are used. The area of occurrence is mapped on the 10 x 10 km EEA grid. A grid cell is considered occupied if one C1 or at least three independent C2 observations of wolves have been documented. Grid cells with only C3 observations (e.g. sightings) or too few C2 are not considered to be occupied by wolves. In such cases, monitoring should ideally be intensified. The monitoring effort varies greatly between the Länder according to their different monitoring structures and the level of experience of people in charge of evaluating wolf signs. People in charge of monitoring in the Länder with large carnivore presence meet once a year for a country-wide assessment. The meetings have been organized by the Federal Agency for Nature Conservation (BfN) since 2009 and will be the basis for the FFH reporting (Reinhardt 2013).

Austria. All wolf signs are entered into a central database and classified according to the refined German SCALP criteria (Kaczensky et al. 2009). Wolf monitoring has been based on genetic monitoring since 2009. Monitoring in Austria is coordinated by the bear conservation advocates. There is close cooperation on a technical level with colleagues from neighbouring countries including cross-border tracking of radio-collared animals and a coordination of genetic monitoring with Switzerland (Kaczensky & Rauer 2013).

Slovenia. In the past, population estimates were based on "integral monitoring" of the population by the employees of the Slovenia Forest Service. The monitoring effort included the opportunistic recording of direct observations, tracks, scats, livestock damages, litter and pup findings and the reconstruction of past mortality. In 2010, a genetic CMR method was used to obtain reliable population size estimates. In the same year, systematic wolf howling was also introduced with the purpose to record presence of territorial wolves and reproduction (Majić Skrbinšek 2013).

4.2. Lynx population status

4.2.1. Present distribution and abundance of the lynx in the Alps

The Alpine lynx consists of five relatively isolated subpopulations (Molinari-Jobin 2010). They are slowly recolonising the Alpine region and their status has been studied in varying detail in the respective countries. Over the past 10 years, they have increased their area of presence by around 6,000 km^2 or 50%, mainly after the translocation in Switzerland (Chapter 3.2.2). However, the population size remained more or less the same (Table 4.1).

In 2011, the number of lynx estimated for the entire Alpine region was between 130 and 180 individuals (Table 4.1), based on reports from the national monitoring projects and a compilation of lynx presence data classified according to the three SCALP categories (Fig 4.2).

France. Although the potential habitat for the lynx in the French Alps is large (~4,300 km², see also Chapter 7.2), it was difficult to accurately determine what percentage of it was occupied (Vandel & Stahl 2005). Confirmed and unconfirmed signs of lynx presence were recorded in the eight Alpine departments of Alpes-de-Haute-Provence, Alpes Maritimes, Drome, Hautes Alpes, Haute Savoie, Isere, Savoie and Var. However, the observation effort was relatively low and the actual presence was hypothesised to be higher than shown (Vandel & Stahl 2005). However, the rather low presence has been confirmed since. The estimated regular population range covered less than 1,350 km² in 2009, which may hardly correspond to more than 10–15 resident adults (Marboutin et al. 2012). Compared to 1999–2007 the distribution was even reduced (ONCFS 2011). The area of lynx presence in the French Alpine departments was mapped out based on regular and recent presence (Fig.4.3).

Table 4.1. Lynx abundance in the Alps in 2001 and in 2011 (mature inidviduals). Sources: von Arx et al. (2004), SCALP (2012), Kaczensky et al. (2013a).

Country	2001	2009–2011	2011**	Trend 2006 –2011
France	single individuals	15–20	13*	West: slight increase
Italy	10-13	10 –15	10-15	
Switzerland	70	100–120	96-107	
Liechtenstein	0	0 –2	0	East: decrease
Germany	0	0 –1	0	
Austria	20	6 –12	3-5	
Slovenia	10	5 –10	few	
Alps	**~120**	**136 –180**	**~130**	

*extrapolated from densities of the Jura population. ** estimates in Kaczensky et al. (2013a).

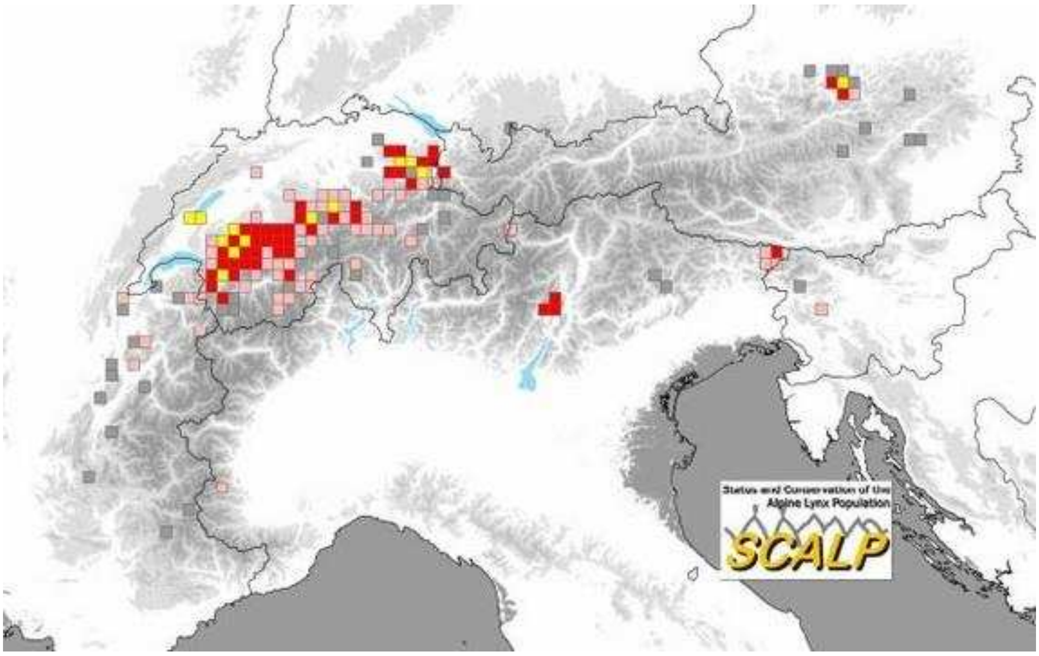


Fig. 4.2. Lynx distribution in the Alps 2012 based on a 10x10 km grid; yellow cells = cells where reproduction was reported from, red cells = C1, cells with hard fact records, pink cells = C2, cells with confirmed records, grey cells = C3, cells with unconfirmed record (SCALP 2012).

Italy. Although reintroduction projects within Italy were not successful, reintroduction projects in Austria and Slovenia contributed to the recolonisation of lynx in the north-eastern regions of Italy in the 1980s (Chapter 3.2.2.; Molinari et al. 2006). Between 2005 and 2009, 268 signs of lynx presence were collected and analysed (Fig. 4.4). However, data show that between 2000 and 2009, the number of signs has decreased and the area of presence has reduced to one-third of the original area thought to be occupied by lynx. Using camera traps and radio-telemetry, four lynx individuals were identified in the Italian Alps in 2009 (Molinari et al. 2012). In April 2014, a male and a female lynx were translocated from Switzerland to Tarvisio in the Julian Alps in order to reinforce the south-eastern Alpine/Dinaric lynx population (Molinari-Jobin 2014).

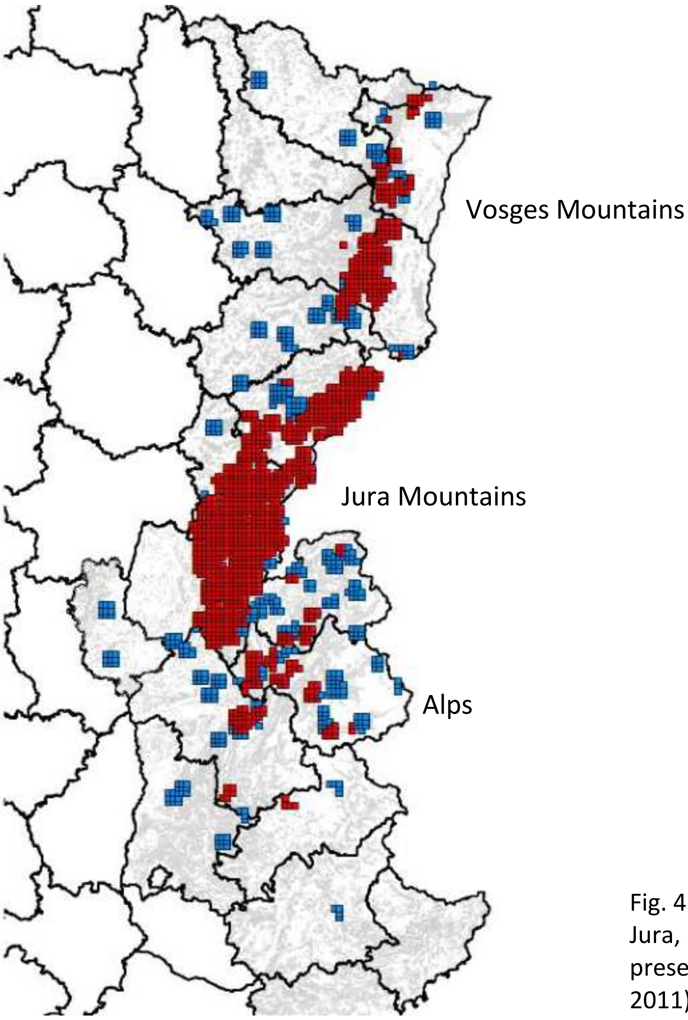


Fig. 4.3. Area of lynx presence in France (Vosges, Jura, and Alps) in 2008–2010. Red = area of regular presence, blue = area of recent presence (ONCFS 2011).

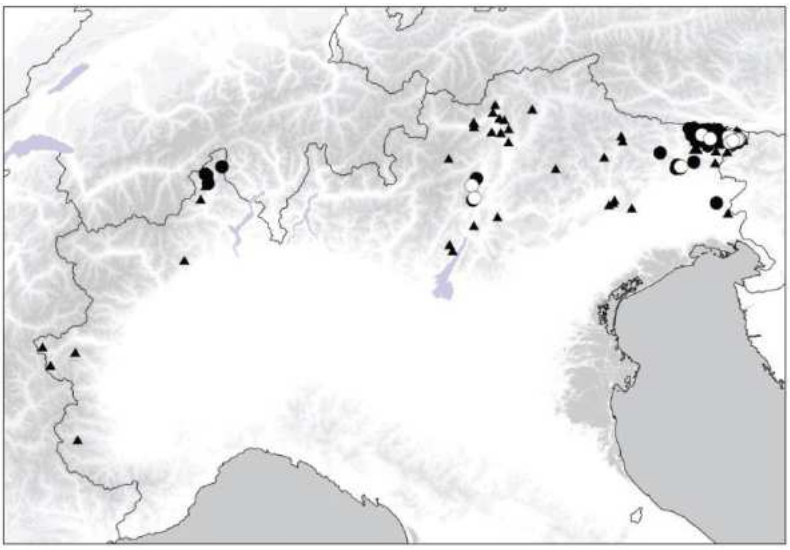


Fig. 4.4. Distribution of lynx signs of presence in the Italian Alps for the five-year period 2005–2009. White points = confirmed hard fact data C1; black points = confirmed data C2; black triangles = unconfirmed data C3 (Molinari et al. 2012).

Switzerland. In the Swiss Alps, the lynx population is part of the western subpopulation and occurs mainly in the north-western and central regions of the country. Between 2001 and 2008, 12 lynx from the north-western Alps and the Jura Mountains were translocated to the north-eastern Swiss Alps to create a new population nucleus and hence to contribute to the expansion of the species especially since spontaneous long distance migrations are rare (Chapter 3.2.1). The population in

Switzerland currently forms the largest subpopulation of lynx in the Alpine region (von Arx & Zimmermann 2013). About 47% of the lynx presence signs were reported from the north-western part of the Swiss Alps (Zimmermann et al. 2011). Signs of reproduction were found in the north-western region between 2000 and 2004 and between 2005 and 2009 with reports of juveniles each year in that region as well as in the north-eastern and central Alps (Fig. 4.5; Zimmermann et al. 2011). Signs of reproduction and mortalities increased in the 2005–2009 pentad in the north-western region.

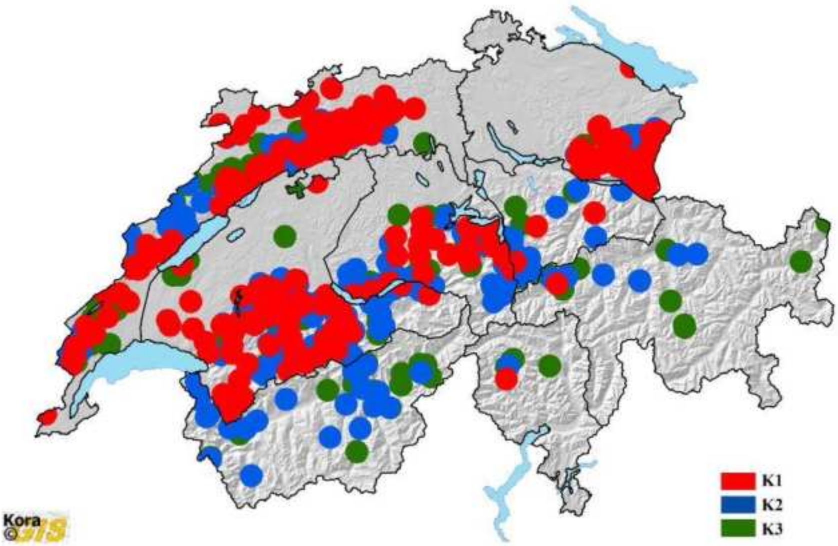


Fig. 4.5. Lynx distribution in Switzerland in 2013, classified according to the SCALP categories and buffered with a 5-km rim. Red= C1 data, blue= C2 data, green= C3 data (KORA 2014).

Liechtenstein. Between 2005 and 2007 there were two records of lynx tracks and in 2008 there was an unconfirmed record of a lynx sighting (Fig. 4.6; Frick 2012). Using the SCALP categories, the tracks were classified as C2 data and the sighting as C3. In March 2013 there was a visit from a female lynx radio-collared in the canton of Grisons (Chapter 3.2.2), however, no lynx settled in the country so far.

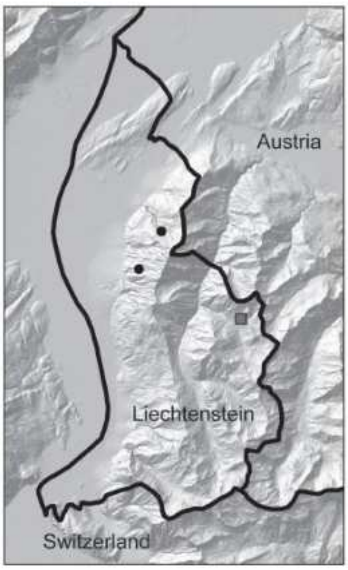


Fig. 4.6. Lynx records between 2005 and 2009 in Liechtenstein. Black dots are C2 data (tracks from 2007 and 2009) and the grey square is aC3 data (sighting in 2008); Frick 2012).

Germany. Germany hosts two lynx populations, the Bohemian–Bavarian population at the border to the Czech Republic and the Harz population which established itself from 2000 on, following the release of captive-born animals (Fig. 4.7). There are no lynx in the German Alps (Chapter 3.2.2). Between 2004 and 2009, there were five sightings all of which were classed as C3 based on the SCALP categories (Wölfl & Wölfl 2011). It remained unclear if these sightings were from a transient dispersing individual or if they were “false positives”.

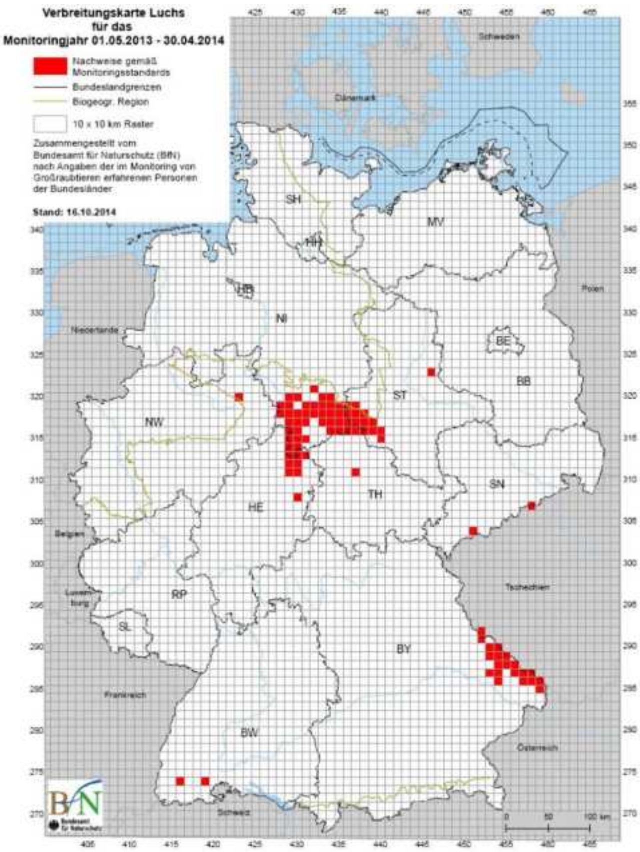


Fig. 4.7. Distribution of lynx in Germany (BfN 2014).

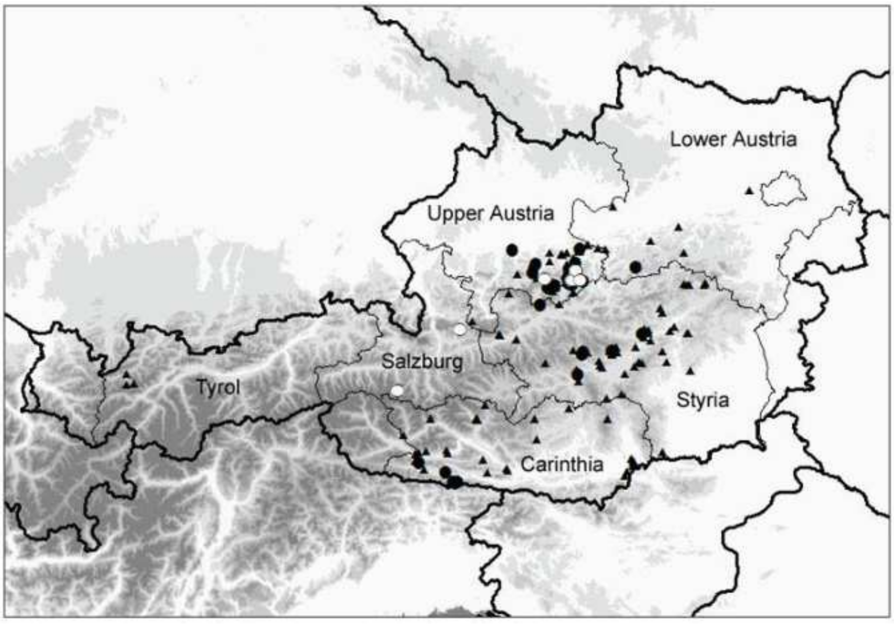


Fig. 4.8. Distribution of lynx signs of presence in the Austrian Alps for the five-year period 2005–2009 (white points = confirmed hard fact data C1; black points = confirmed data C2; black triangles = unconfirmed data C3); (Fuxjäger et al. 2012).

Austria. Nine wild lynx from the Carpathian Mountains were reintroduced in Styria between 1977 and 1979 (Huber & Kaczensky 1998), but the reintroduction failed. Recent observations concern lynx that immigrated from the Dinaric population or the Bavarian-Bohemian population (in the east) or from eastern Switzerland (in the west; Chapter 3.2). The population in the Kalkalpen National Park consisted of a male which settled in the area in 1998 (Kaczensky et al. 2013c). A female from the Swiss Alps and a female and a male from the Jura Mountains were translocated to the Kalkalpen National Park in 2011 and 2013; both females have already reproduced (Fuxjäger 2014; Nationalpark Kalkalpen 2011a, b). According to Kaczensky et al. (2013c), there may be additional 1–2 animals elsewhere in the Austrian Alps. Assumed occurrences have so far not been confirmed (Kaczensky et al. 2013c). Between January 2005 and December 2009, 228 records of lynx were collected (Fig. 4.8; Fuxjäger et al. 2012). Of these, 61% (140 samples) were classified as C3 data, and 14% as C1 data consisting of 32 camera-trap photos. Of the 32 C1 data, 30 showed a single individual from the Kalkalpen National Park, which was pictured the first time in 2000. The other two camera-trap photos were taken in 2009 by hunters of two individuals in two different regions of Salzburg, from where no other signs (neither C2 nor C3 data) were known (Fuxjäger et al. 2012).

Slovenia. The Slovenian lynx population is divided between the Dinaric and the Alpine populations (Kos & Potočnik 2013). The Jesenice – Ljubljana – Trieste highway marks the border between the two populations. The majority of the Slovenian lynx belong to the Dinaric population (Fig 4.9).

A substantial reduction in the population has been indicated since 2005, based on reports of an absence of lynx signs in areas which had previously been occupied by lynx. Estimates place the current population at 15–25 individuals, with about 5–10 individuals in the Alpine region and 10–15 in the Dinaric Mountains in southern Slovenia (Kos & Potočnik 2013). However, this estimate is not based on a robust methodology as the monitoring was opportunistic in nature. A net decrease in the ungulate populations, notably red deer and roe deer, appeared to have led to a decrease in the lynx population (Krofel 2005; see Chapter 5.3). However, not all data support such a decrease in prey populations (M. Jonozovič, pers. comm.). Other possible causes include inbreeding depression, illegal hunting and the expanding wolf population. The Dinaric population is highly inbred (Chapter 4.2.2), and the lack of observations of reproduction in recent years supports the hypothesis of a possible inbreeding depression.

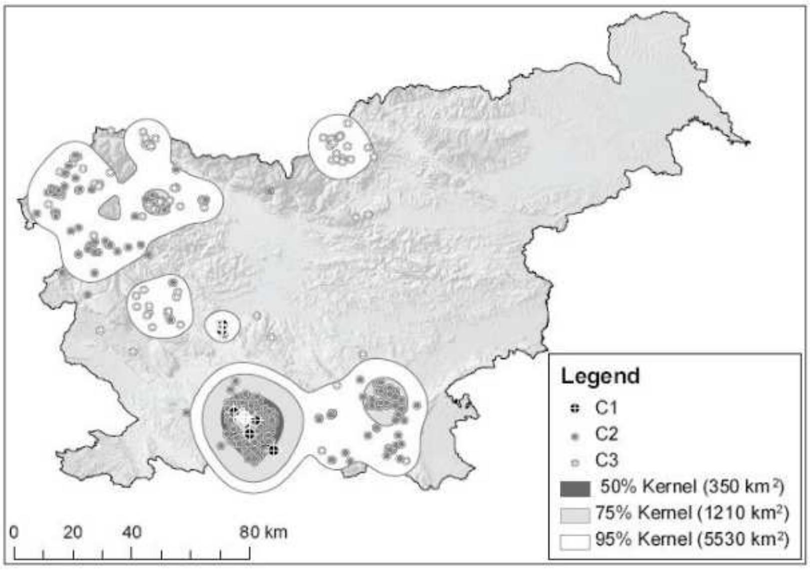


Fig 4.9. Lynx population range in Slovenia 2005-2009 (Kos et al. 2012).

4.2.2. Assessment of the Alpine lynx population

Box 4.1. Genetic Analyses: Glossary

Allele: An allele is one of two or more forms of the DNA sequence of a particular gene. The word is a short form of allelomorph ('other form'), which was used in the early days of genetics to describe variant forms of a gene detected as different phenotypes.

Allelic diversity: A measure of genetic diversity within a population, computed as the average number of alleles per locus.

Bottleneck: A special case of strong genetic drift where a population experiences a loss of genetic variation by temporarily going through a marked reduction in effective population size. The bottleneck can last from one or a few generations, to one of much longer duration. The longer the bottleneck lasts the more severely the population is affected.

Effective population size N_e: The average number of individuals in a population that contribute genes to succeeding generations.

Expected heterozygosity H_E: The heterozygosity expected for a random mating population with the given allele frequencies according to the Hardy-Weinberg equilibrium.

Factorial component analysis: Regression models predicting observed variables by latent variables.

Genetic drift: Changes in the genetic composition of a population due to random sampling in small populations. Results in loss of genetic diversity, random changes in allele frequencies, and diversification among replicate populations. While drift occurs in all populations, its effects are most evident in small populations.

Haplotype: Variation of a DNA-sequence on a single chromosome of an individual. A certain haplotype can be particular to an individual, a population or a species.

Heterozygote: An individual with two different alleles at a locus, e.g. A_1A_2.

Homozygote: An individual with two copies of the same allele at a gene locus, e.g. A_1A_1.

Inbreeding: The mating between related individuals resulting in an increase of homozygosity in the progeny because they get alleles that are identical by descent.

Inbreeding coefficient: The most commonly used measure of the extent of inbreeding; the probability that two alleles at a locus in an individual are identical because they come from a common ancestor. Ranges from 0 to 1.

Inbreeding depression: Decrease in health and fitness often observed in offspring resulting from inbreeding.

Locus: A segment of DNA on a chromosome.

Microsatellite: also known as short tandem repeats (STRs), are repeating sequences of 2-5 base pairs of DNA. Microsatellites are typically co-dominant. They are used as molecular markers in STR analysis, for kinship, population and other studies.

Major Histocompatibility complex (MHC): A large family of loci that play an important role in the vertebrate immune system and in fighting disease. These loci show extraordinarily high levels of genetic diversity.

Mitochondrial DNA (mtDNA): The circular DNA molecule contained within mitochondria. Maternally inherited.

Polymorphic: The presence of more than one allele at a locus.

Observed heterozygosity H_O: The actual level of heterozygosity measured in a population.

Single nucleotide polymorphism (SNP): A nucleotide site (base pair) in a DNA sequence that is polymorphic in a population and can be used as a marker to assess genetic variation within and among populations.

IUCN Red List assessment

The total population of lynx in the Alps is still small and **Endangered** according to the IUCN Red List assessment (IUCN Standards and Petitions Subcommittee 2014). Kaczensky et al. (2013a) have assessed the large carnivore populations across Europe, based on the most recent data available. They have listed the Alpine lynx population as EN(D), meaning that the population is Endangered under the criterion D (total population size smaller than 250 mature individuals). The population was considered stable or slightly increasing in Switzerland, and stagnant in Italy, France, Austria, and Slovenia, and the conclusion was that "the observed rate of development will most likely not allow for a natural fusion of the western and eastern Alpine populations within the next decades". Boitani et al. (2015) listed as most relevant threats to Eurasian lynx low acceptance due to conflicts with hunters, persecution and habitat loss due to infrastructure development, poor management structures and accidental mortality (Chapter 4.4). Additionally, inbreeding is listed as an important threat to the Alpine population by Kaczensky et al. (2013a; Appendix II).

Genetic viability of the present populations

The loss of genetic diversity through a high level of inbreeding can lead to a reduced fitness called inbreeding depression (Keller & Waller 2002, Box 4.1). The correlates vary greatly and are often hard to detect: reproductive problems (lower sperm quality and quantity, reduced litter size), increased disease susceptibility (e.g. for Feline Leukemia Virus), malformations (heart, skeleton, skin, cryptorchidism) or histological lesions (kidneys, heart; Breitenmoser 2011). The reintroduced lynx populations in the Swiss Alps, the Jura Mountains and in the Dinaric range have been monitored for changes in genetic variability for the past 30 years.

Inbreeding results in an increase of homozygosity and, correspondingly, a decrease of heterozygosity in the progeny because they have alleles that are identical by descent (Allendorf & Luikart 2007). Breitenmoser-Würsten & Obexer-Ruff (pers. comm.) performed an analysis of 22 microsatellites in eleven populations in Europe, both autochthonous and re-introduced. They found low levels of heterozygosity in the populations in the Swiss Alps and the Dinaric range with values below 0.5 and low mean numbers of alleles per locus of less than 3.0. Both parameters were considerably higher in the source population in the Carpathian Mountains of Slovakia and other large outbred populations (He 0.6-0.7; >4.0 Alleles/locus). The population in the Swiss Alps also experienced a strong genetic drift, with loss of rare alleles and changes in allele frequencies. The genetic drift was strong enough that today the population in the Swiss Alps is clearly distinct from its source population as can be illustrated with a factorial component analysis FCA (Fig 4.10; Breitenmoser-Würsten & Obexer-Ruff 2003).

The Swiss Alpine population manifested six of the seven known cases of congenital malformations in Switzerland, with only one found in the Swiss Jura Mountains (Ryser-Degiorgis et al. 2004). The study was based on necropsies, veterinary examinations of orphans brought to a wildlife rescue centre, and veterinary examinations of animals captured for radio-collaring and/or translocations. How these malformations relate to the genetic impoverishment is not yet clear. However, in comparison, only in one of over 500 necropsies a skeletal disorder was detected in lynx in Sweden (Ryser-Degiorgis et al. 2004). Additionally, the amount of malformations in the Swiss population might be underestimated: animals that die of diseases or starvation are less likely to be found in field studies than animals dying from human causes such as traffic accidents or hunting (Schmidt-Posthaus et al. 2002, Ryser-Degiorgis et al. 2004).

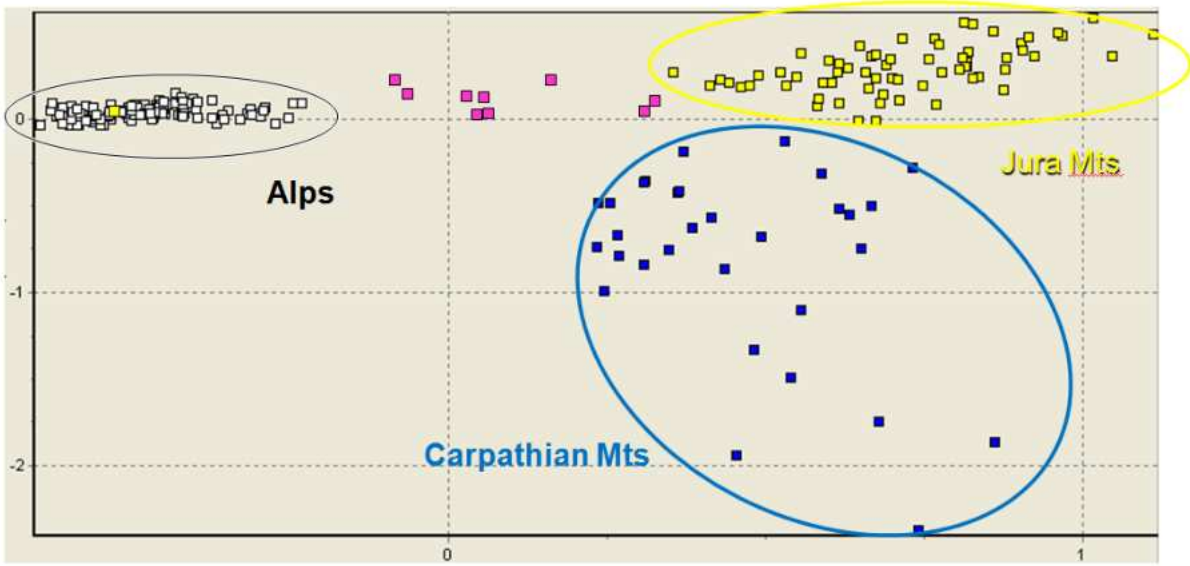


Fig. 4.10. Factorial Component Analysis based on allele frequencies of lynx from the source population (blue), the population in the Swiss Alps (white) and the Jura Mountains (yellow). Each pixel represents one individual of the respective population. The ellipses are hand drawn and have no statistical value. The pink symbols are offspring from a mating of a male from the Alps with a female from the Jura Mountains (Breitenmoser-Würsten, pers.comm.).

The populations in the Jura Mountains, Bavarian/Bohemian Forest and Dinaric Mountains have all been founded with animals from the Carpathian Mountains as well. The population of the Jura Mountains also showed a similar level of differentiation from the Carpathian source population as the one from the Alps but the drift went into a different direction (Fig 4.10). The two populations within Switzerland are therefore today distinct. The other re-introduced populations drifted less away from the source population; they were founded with a larger number of individuals that were released over a longer period of time, thus less likely closely related animals have been released (Breitenmoser-Würsten & Obexer-Ruff 2003).

The population of the Dinaric Mountains extends into the south-eastern Alps of Slovenia and is a potential source for the recolonisation of the Alps from the east (Chapter 4.2). However, the genetic situation of this population is also challenging. The population was founded with six individuals, including one mother-son pair, and a brother-sister pair (Koubek & Cerveny 1996). Sindičić et al. (2013) also found a very low level of genetic diversity compared to other European populations. The analysis was done on samples collected between 1979 and 2010 from Bosnia & Herzegovina, Croatia and Slovenia, including a few samples originating north of the Ljubljana-Trieste highway, which is considered to be the border between the Dinaric and Alpine populations. Expected and observed heterozygosity have shown a decrease since 1991 (Fig. 4.11). The effective population size was calculated using two methods. The first method (ONeSAMP) resulted in a stable, slowly increasing effective population size of 14.8 in 1979–1990, 16.0 in 1991–2000 and 16.2 in 2001–2010. The other method (LDNe) is more sensitive to rapid changes and showed a sharp drop in the effective population size after the year 2000 from about 22 down to 10.9 (Sindičić et al. 2013). The effective inbreeding coefficient was increasing and reached F = 0.3 in the 1999–2010 period. In a recent workshop, an early warning system and a threshold for conservation actions to counteract an inbreeding depression were suggested:

> "(1) Fit is approaching 0.25 (every population member is a sibling of each other; this value needs to be compared to outbred populations as it is relative); (2) Ne (genetic effective population size) is below 50. This value can be for a single population or a metapopulation" (Breitenmoser 2011).

A rigorous monitoring of both numbers N_b and the genetic effective population size N_e is necessary. Demographic estimates can be up to 10 times higher than corresponding genetic estimates, and caution is needed with high numbers masking a high level of inbreeding.

A serious inbreeding depression of F = 0.47 with multiple correlates was found in the Florida panther (*Puma concolor*), a value which could be reached in 12–17 generations (48–68 years) in the Dinaric lynx population (Sindičić et al. 2013). Such an interspecific comparison and extrapolation is only indicative, but hints that "the Dinaric lynx population will reach a point where inbreeding depression becomes a real problem sooner rather than later" (Sindičić et al. 2013).

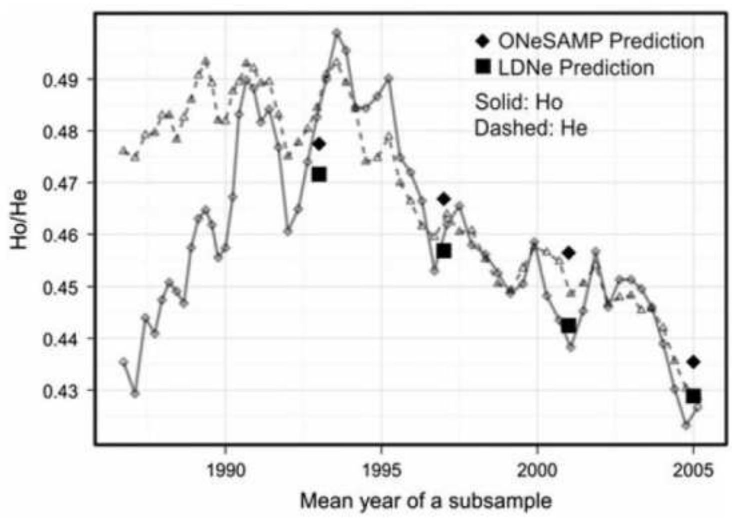


Fig. 4.11. Mean observed (H_O) and expected (H_E) heterozygosity of the 30-samples travelling window subsamples. Black diamonds: predicted He using LDNe estimates of the effective population size, black squares: predicted H_E using ONeSAMP estimates of N_e (Sindičić et al. 2013).

Although the genetic situation of the reintroduced populations in and around the Alps is quite alarming, there are remedies. Firstly, the release of a few individuals from the source population could improve the genetic situation. The release of eight individuals in the aforementioned population of Florida panther had huge positive effects (Johnson et al. 2010). Secondly, connecting the (sub-) populations would result in a larger effective population size with increased genetic diversity due to the genetic exchange between the formerly separated populations. Such a connection might benefit from management actions such as translocation. So far it was possible to document one male from the Swiss Alps that managed to reach the Jura Mountains where it reproduced successfully (Fig 4.10). Two other males have been translocated as a management intervention. The injection of new genetic information led to a raise in heterozygosity and mean number of alleles per locus (C. Breitenmoser-Würsten, pers. comm.).

4.3. Wolf population status

4.3.1. Present distribution and abundance of the wolf in the Alps

The border between the wolf population in the Italian Apennine and the population in the south-western Alps has been arbitrarily set at the Colle di Cadibona, which also marks the geographical separation between the Alps and the Apennines. Although there is evidence of a continuous, albeit tenuous genetic flow between the two populations (Fabbri et al. 2007, Ciucci et al. 2009) and, by all biological standards, they form only one population, practical considerations and important differences in the ecological, social and economic contexts justify distinguishing two populations for management purposes (WAG 2011).

The wolf abundance was estimated to at least 160 individuals in 2009/2010 (Table 4.2). As the Wolf Alpine Group considers changes in the number of "wolf packs" as the biologically meaningful measure of population trend and distribution, they compiled data to create a map of the distribution of packs (Fig. 4.12) and a graph showing the evolution of the number of wolf packs and pairs across the Alpine countries between 1992 and 2012 (Fig. 4.13). The trend was almost steadily increasing during this time.

Table 4.2. Wolf abundance in the Alps in 2009/10 (Kaczensky et al. 2013a).

Country	**Minimum number**	**Number of packs**
France	68	13 packs + 7 transboundary packs [2009/10]
Italy	67	12 packs + 7 transboundary packs [2009/10]
Switzerland	8 [2011]	1 pack [first reproduction in 2012]
Liechtenstein*	no wolves	
Germany*	occasional dispersers	none in the Alps
Austria	2-8 [2009-2011]	none
Slovenia	occasional dispersers	none in the Alps
Alps	**>160 wolves**	**32 packs [2009/10]**

*Not included in Kaczensky et al. 2013a.

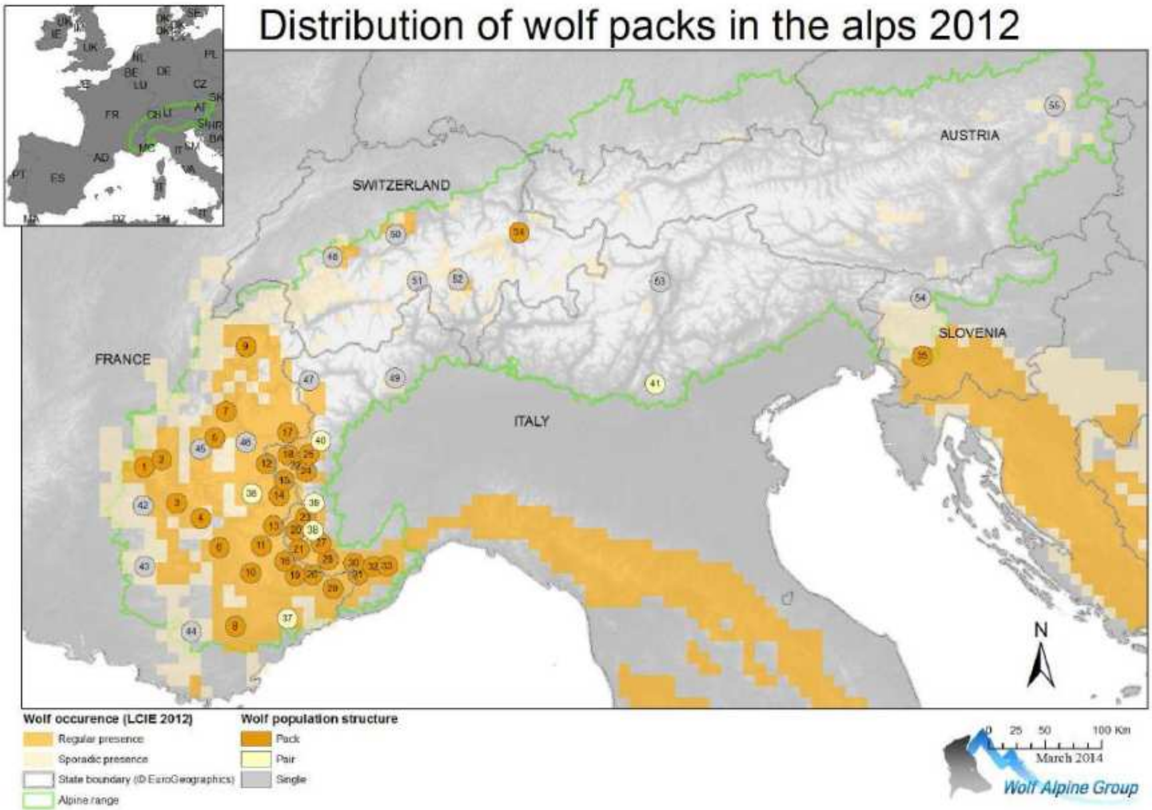


Fig. 4.12. Distribution of packs, pairs and single wolves in 2012 in the Alps that hold a territory for at least two years (WAG 2014).

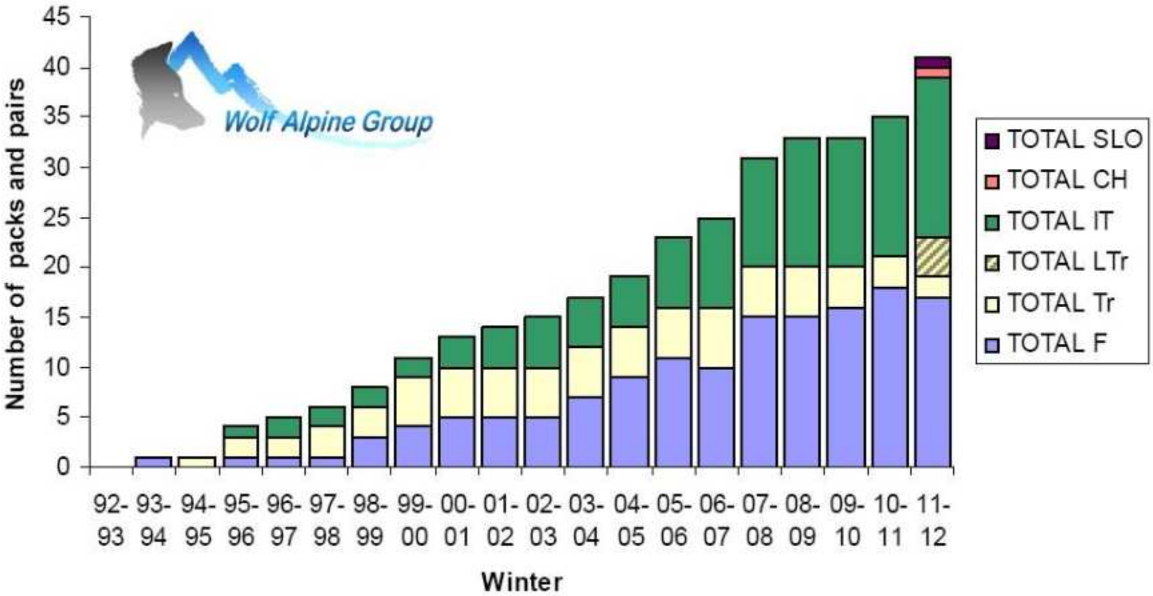


Fig. 4.13. Number of wolf packs and pairs across the Alpine countries (WAG 2014). NB: packs and pairs = at least 1M + 1F for two consecutive winters or breeding evidence next summer. F: France; IT: Italy, CH: Switzerland; SLO: Slovenia, Tr: transboundary, LTr: likely transboundary.

France. In France, an evaluation of the population status is carried out in spring each year to determine if the population has increased or decreased (ONCFS 2014a). Wolves in France are mostly found in the Alpine region (Marboutin 2013a). Based on snow tracking, a population consisting of 43 wolves was estimated in 2004/2005 (Salvatori & Linnell 2005). Census results in 2009 resulted in the identification of 13 wolf pack territories and 7 transboundary pack territories straddling France and Italy (Marboutin 2013a). By the 2009/2010 season, the population estimated through snow tracking was around 68 wolves. Results of a wolf presence study in 2014, in the whole of the country show a 14% increase of regular presence and a 17% increase of occasional presence (Fig 4.14; ONCFS 2014a). Occasional presence in this case refers to less than 3 presence signs over a 2-years moving window (Marboutin & Duchamp 2005).

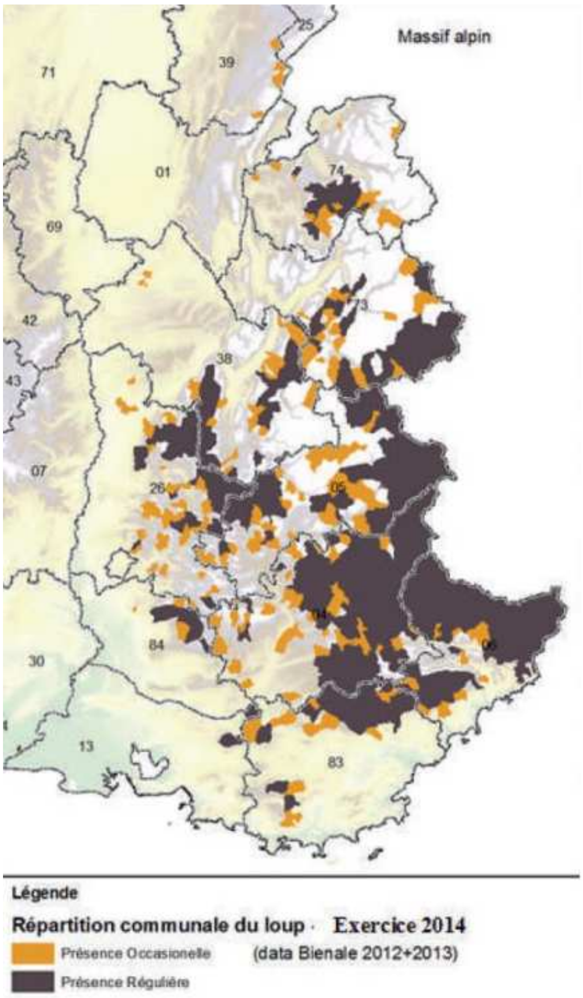


Fig 4.14. Areas of regular (dark brown) and occasional (light brown) presence of wolves in the French Alpine region (ONCFS 2014a).

Italy. In 1992, the wolf reappeared in the western Alps, following an expansion of the peninsular population, and a new (Alpine) population has been steadily occupying both the Italian and French side of the Alps ever since (Chapter 3.3.1; Boitani & Marucco 2013). Wolf pack territories in the Piedmont region were studied in 2010–2011 and mapped (Fig 4.15). The population in the Italian Alps was estimated to around 60-70 wolves (Marucco & Avanzinelli 2012), distributed across at least 12 packs, in addition to seven transboundary packs occurring between Italy and France. Of the 12 packs, 11 packs were found in the Piedmont region and one in Val d'Aosta (Gran Paradiso National Park) (WAG 2011). The current population on the Italian side is spread across 5,500 km^2 and dispersing animals have been found as far as the western Pyrenees, southern Germany and Austria[4], eastern Switzerland as well as in the central–eastern Italian Alps (Boitani & Marucco 2013). The Italian and Dinaric wolf populations were separated for centuries. In 2012, the first contact between these two populations was documented in the eastern part of the Italian Alps (Veneto region) when a male wolf from Slovenia and a female wolf from the Italian Alpine population formed the first breeding pair in the area (Boitani & Marucco 2013).

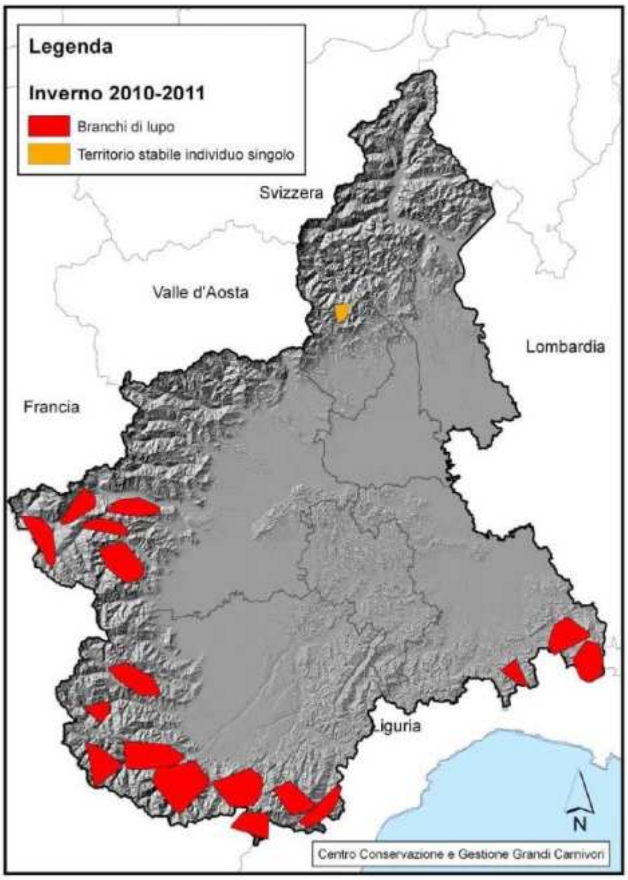


Fig. 4.15. Minimum area of the territories of wolf packs in the Piedmont region in the winter of 2010–2011 (Marucco & Avanzinelli 2012).

Switzerland. A total of 60 individuals were genetically identified from 2005–2014, among which 14 females. The number of identifiable animals fluctuated considerably over the years. So far, most of the individuals were only detected over one or two years and then disappeared. To date 15 dead wolves have been found in Switzerland. All died due to human causes (KORA 2014). In 2011, three individuals were considered resident as they had been present in an area for three years. The first evidence of wolf reproduction in Switzerland was confirmed in August 2012, in the region of Calanda (canton of Grisons) (von Arx & Manz 2013). This pack has reproduced in 2012, 2013, and 2014, but no other pack has established so far. From 01.10.2012 to 30.09.2014 24 wolves (17 males and 7 females) could genetically be identified. (Fig. 4.16; KORA 2014).

[4] It is not proven that a wolf from the Italian Alps dispersed to Austria. So far, 4 individuals with the Italian haplotype W1 (Valière et al. 2003) have been identified in Austria. One originated from the French Alps and one probably from the Calanda pack in Switzerland. The origin of the other two could not be established more exactly than Alps, i.e. Italy/France/Switzerland (G. Rauer, pers. comm.).

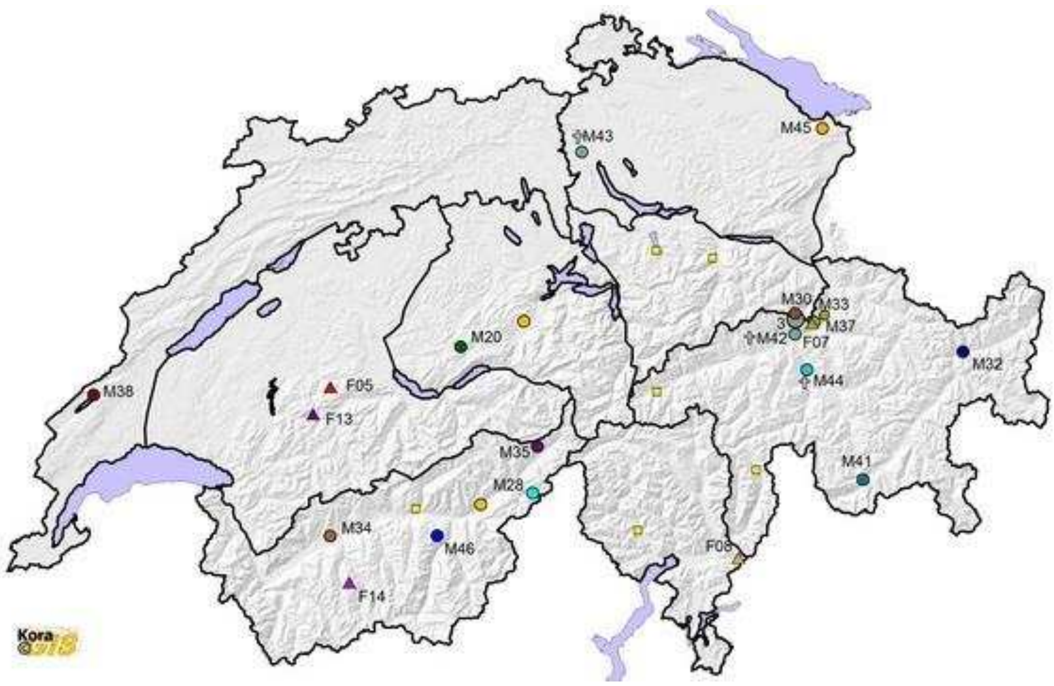


Fig. 4.16. Wolf evidence in Switzerland October 2012 till September 2014. Genetically individually identified wolves are indicated with a symbol and a sequential number (M = Male, F = female, yellow squares = unknown individuals (not possible to identify or not yet analysed). The genetic analyses are conducted at the Laboratory for Biology and Conservation from the University of Lausanne (KORA 2014).

Germany. Wolves are mainly present in the north-eastern part of Germany (Central-European Lowlands population in Fig. 3.13). So far, there was no evidence for an expansion of the population towards the south and south–western parts of the present distribution range (Kaczensky et al. 2008, Reinhardt 2013). However, three lone wolves were identified in Bavaria: in 2006, a young dispersing male from the south-western Alps was run over near Starnberg, from December 2009 to January 2011, a male wolf originating from a pack south-west of Mont Blanc, France, roamed the area of the Mangfall Mountains (and according to G. Rauer (pers. comm.) was also detected across the border in Austria), and in autumn 2011, an individual of German/western Poland origin was identified (Bayerisches Landesamt für Umwelt 2014a). In spring 2014, two different male wolves were identified in the Bavarian Alps (press releases of the Bavarian State Office for Environment on 16th April 2014 and 11th July 2014).

Austria. Dispersing individuals originating from the (Italian) Alpine, Eastern Europe and Dinaric-Balkan populations have been genetically identified in different parts of the Austrian Alps (Fig. 4.17). As in Switzerland, there was also a high turnover in the wolf population in Austria. Most wolves were detected only once or a few times within a single year before disappearing again. The frequency of wolf visits to Austria increased slightly over the past 15 years, and both males and females were identified (Rauer et al. 2013). In 2009, the situation changed drastically as 6–8 individuals were genetically identified. The same number but with slightly different individuals was confirmed in 2010. In 2011, 2–3 wolves were identified. As many samples could only be tested for their mitochondrial DNA and an individual genotyping was not possible, the exact number and the duration of stay of single individuals were often not determined. In the winter of 2011/2012 a radio–collared wolf from Slovenia crossed Austria on its way to the Italian Alps and remained in the country for a period of 38 days (Kaczensky & Rauer 2013). One wolf, a male from the Italian-Alpine population, had settled for about two years in the area of Schneeberg in Lower Austria but seemed to have disappeared by 2013 (Rauer et al. 2013). Its disappearance was ascertained in the meantime (G. Rauer, pers. comm.). Two more stationary individuals (duration >1 year) have been known in Austria so far: one was killed in May 2014 in a barn, when it was mistaken for a fox; the other one was lastly detected in August 2014 and might still be present today (G. Rauer, pers. comm.). No other stationary wolves were observed in Austria so far. Nevertheless, further immigrations and the establishment of packs in the country are expected in the near future (KOST 2012).

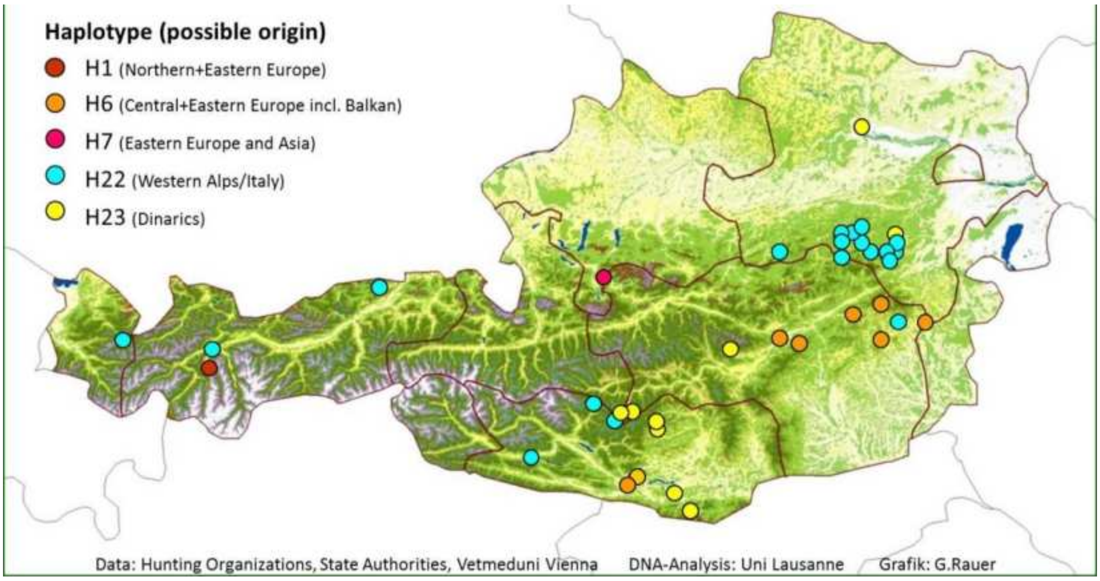


Fig. 4.17. Evidence of wolves in Austria 2009–2012 illustrating that wolves from different European populations are immigrating into the Austrian Alps (WAG 2014).

Slovenia. Wolf distribution in Slovenia represents the north-western part of the Dinaric-Balkan wolf population (Fig. 3.13). They are distributed in south-western Slovenia (Dinaric Mountain chain), along the border with Croatia, towards the coast and in Trnovo forest in the North (Majić Skrbinšek 2013). According to the map (Fig. 4.18), there is only a sporadic occurrence in northern Slovenia, along the southern rim of the Alps. There is a well documented case of a male that dispersed in 2011 from southern Slovenia to the Italian Alps and founded a pack with a female from the Italian population (Fig. 7.4 in Chapter 7.1). In 2010, a genetic CMR method was used to obtain reliable population size estimates. The estimate was calculated using a genetic mark-recapture study for the entire wolf range in Slovenia as well as transboundary packs with Croatia. After correcting for the transboundary wolf packs (which were divided in half), the maximum estimate in September 2010, after reproduction and before the cull, was 43 wolves for Slovenia. The minimum estimate in May 2011, after the cull and before reproduction was 32 wolves (Majić Skrbinšek 2013). Wolf packs in Slovenia were studied and area of wolf presence mapped by Potocnik in Marucco et al. (2013; Fig. 4.18).

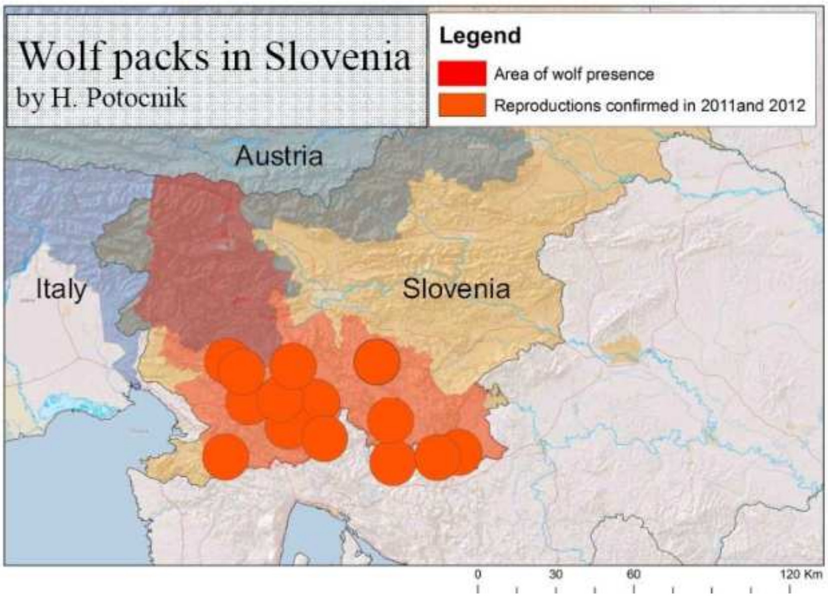


Fig. 4.18. Wolf packs and area of wolf presence in Slovenia 2012 (Potocnik in Marucco et al. 2013).

4.3.2. Assessment of the Alpine wolf population

IUCN Red List assessment

The Alpine wolf population was also assessed as EN (**Endangered**, no criterion, but obviously due to the still low population size of less than 250 mature individuals), but with an increasing population trend. The most relevant threats Boitani et al. (2015) listed are low acceptance, habitat loss due to infrastructure development, persecution, hybridisation with dogs, poor management structures and accidental mortality (Chapter 4.4; Appendix II).

Genetic viability of the present populations

The western Alps have been recolonised by wolves from the Italian population which had experienced a bottleneck and was reduced to about 100 individuals in the 1970s (Zimen & Boitani 1975, Chapter 3). Even though the last wolves in the Alps were only eradicated towards the end of the 19th century (Chapter 3), it was speculated that the Italian wolf population in the Apennines may have been isolated for thousands of years as a consequence of natural landscape changes caused by the last Pleistocene glaciation (Lucchini et al. 2004). More recently, the isolation of the Apennine population from the former Alpine population was maintained by deforestation and simultaneous eradication of wild ungulates (Sereni 1961 in Lucchini et al. 2004). This long-lasting isolation has led to the development of a unique haplotype in the Apennine population which has so far been found in every individual of this population and nowhere else (Randi et al. 2000). This unique haplotype has enabled scientists to attribute the Italian origin of the colonisers of the Western Alps.

Italian wolves are also morphologically distinct from other wolves. Nowak & Federoff (2002) have therefore suggested them to be a distinct subspecies *Canis lupus italicus* based on skull measurements. These morphological findings are also supported by results from genetic analyses. VonHoldt et al. (2011) analysed a dataset of 43,953 single nucleotide polymorphisms SNPs in wolf-like canids, using the software STRUCTURE. The total sample was first split into two groups, separating dogs from wild canids, then coyotes from wolves, Old World from New World wolves, and finally as a first geographically distinct population, Italian wolves from the rest. A similar analysis was performed by Pilot et al. (2013) with 33,958 SNPs from 103 wolves from around the world and five coyotes as outgroup.

They used STRUCTURE and ADMIXTURE, for the same analysis and both recognized the Italian wolves as distinct population. This was also confirmed by a principal component analysis PCA (Pilot et al. 2013). The same conclusion emerged from the PCA performed by Stronen et al. (2013), with data of 67,784 SNPs in European wolves: Italian wolves are the most distinct population (Fig. 4.19).

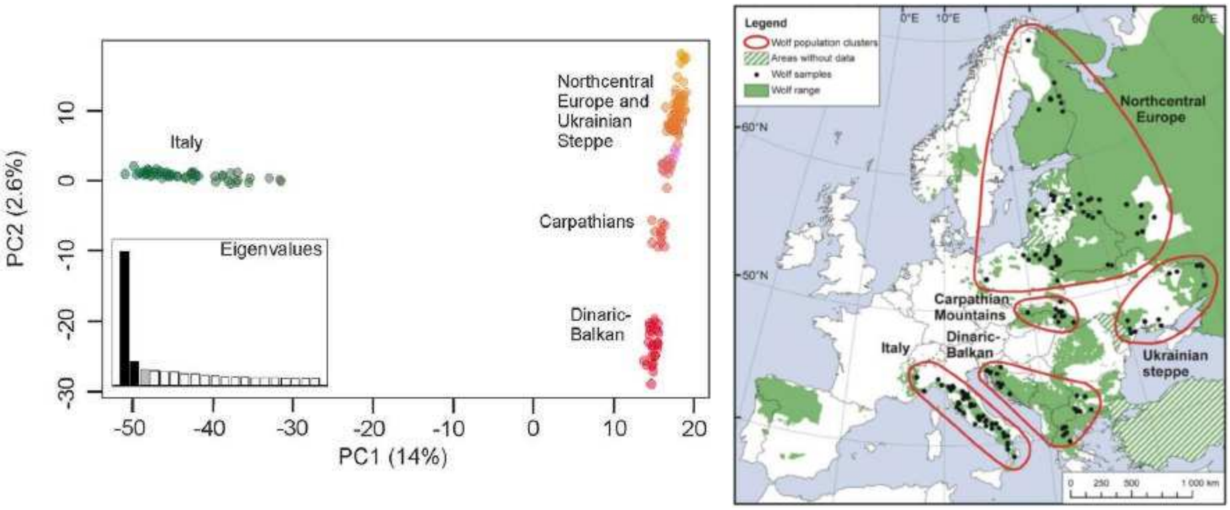


Fig. 4.19. Left: Principal component analysis of European wolves using 67K single nucleotide polymorphism markers SNPs. Genetic similarity is represented by similar colours and spatial proximity (Stronen et al. 2013). Right: Population clusters according to Stronen et al. (2013).

The long-lasting isolation of the Italian population has not only led to its distinction, but also to a decrease in genetic diversity. Heterozygosity and number of alleles per locus are both quite low in the Italian population. Lucchini et al. (2004) found the lowest observed heterozygosity in Europe in Italian wolves and did not find any private alleles in the Italian population, contrary to all other studied populations (2–6 private alleles). The only other populations with such low numbers of alleles per locus were in samples from Saudi Arabia and Turkey–Israel. Both of these populations were represented by a small sample size and a positive but not significant correlation between sample size and number of alleles per locus was found (Lucchini et al. 2004). Apart from the lowest heterozygosity in Europe, the study by Stronen et al. (2013) also found a lower percentage of polymorphic loci (IT: 83.79%, EU: 99.95%) in Italian wolves. Fabbri et al. (2014) found a higher heterozygosity and on average two alleles more per locus in the Croatian population compared to the Italian one. Additionally, they found eleven different haplotypes for four Y-linked autosomal microsatellites in 70 samples from Croatia, and only three in 145 samples from Italy (Fabbri et al. 2014). The study by vonHoldt et al. (2011) based on SNPs also found only a lower heterozygosity in Mexican wolves, worldwide (Table 4.3).However, inbreeding, and consequently an inbreeding depression, do not seem to be a problem in the wolf population. None of the various studies would have called the genetic situation especially worrying or even alarming in terms of conservation. Even though the population is still not very large, Caniglia et al. (2014) found only one of 34 breeding pairs to consist of related individuals (brother and sister) in their study covering over 19,000 km^2 of the Northern Apennines.

Table 4.3. Average observed (HO) and expected (HE) heterozygosity for 48K single nucleotide polymorphism markers SNPs in populations of differing demographic histories (n = sample size, for original sources see von-Holdt et al. 2011).

Demographic history	Population	N	H_O (H_E)
Old World grey wolf			
Large (recent) expanding population	Europe[1]	57	0.24 (0.26)
Historic population bottleneck	Italy	20	0.15 (0.17)
Recent population bottleneck and subsequent expansion with continual hunting pressures	Poland and Belarus	15	0.24 (0.25)
	Russia	18	0.25 (0.26)
Recent population bottleneck	Spain	10	0.18 (0.17)
North American wolf			
Founding from a large source population	Yellowstone National Park	18	0.22 (0.22)
Large constant population size	Canada	13	0.22 (0.24)
Large (recent) expanding population	Western	60	0.21 (0.29)
Recent population bottleneck and subsequent expansion	Minnesota and Southern Quebec	12	0.19 (0.22)
Recent range expansion and potential hybridization	Great Lakes[2]	23	0.18 (0.21)
Recent population bottleneck with managed breeding; possible hybrid-species origin	Red wolf	12	0.16 (0.16)
Recent population bottleneck with managed breeding	Mexican wolf	10	0.12 (0.18)

[1]Excludes Italian and Spanish wolves. [2]Excludes Minnesota and Quebec wolves.

The Apennine population, with its already slightly decreased genetic diversity, was the source for the Alpine population. The unique haplotype of the Italian wolves was also found in all wolves of the new Alpine population in Italy, France and Switzerland (Fabbri et al. 2007). However, the Alpine population is genetically distinct (Fabbri et al. 2007). A range expansion can be viewed as another bottleneck as not all individuals and current genetic information will be represented in the expansion and founding of the new (sub-) population. In the case of the Alps, 8–16 effective founders would explain the genetic diversity found in the population in the Alps (Fig. 4.20, Fabbri et al. 2007). Similarly, other studies found lower genetic diversity in the population in the Alps than in the source population of the Apennines. However, the bottleneck found in the new population was not severe: about 66% of allelic richness and 90% of expected heterozygosity were still maintained after 16 years, equalling 4–5 generations (Fabbri et al. 2007). This is due to a continuing migration of individuals from the source population in the Apennines into the Alps, meaning 1.25–2.5 effective migrants per generation (Fabbri et al. 2007). Lucchini et al. (2002) report a lower heterozygosity and a lower number of alleles in the Alps than in the Apennines. The same results were found by Fabbri et al. (2007) who additionally found seven private alleles in the Apennine population which were not present in the Alpine population. Galaverni et al. (2013) found about half of the observed and expected number of haplotypes in the major histocompatibility complex MHC in the Alps compared to the Apennine population. Finally, of the three haplotypes found in four Y-linked autosomal microsatellites in Italian wolves, only one occurred in the Alps (Fabbri et al. 2014).

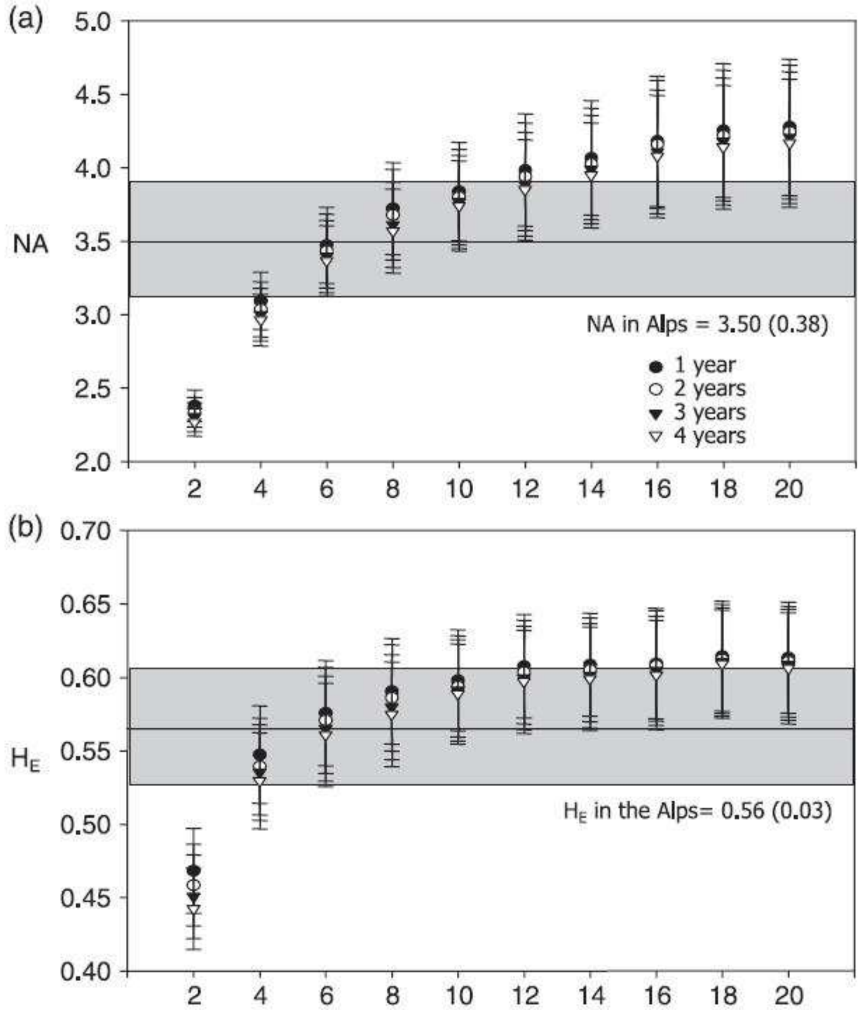


Fig. 4.20. Bottleneck simulation. Effects of bottlenecks of variable size (from a minimum of 2 up to 20 founders) a: on the average number of alleles per locus (N_A); b: on the expected heterozygosity (H_E), in newly founded populations that remain stable for 1–4 years. Dark bars indicate the standard errors of the observed values in the Alpine wolves (Fabbri et al. 2007).

The recolonisation of the Eastern Alps is not as advanced as in the Western Alps, but some pioneers have started to appear there. Contrary to the Western Alps – where only individuals from the Italian population participated in the recolonisation – the pioneers in the Eastern Alps came from various source populations. Haplotypes, which were previously only known from specific regions/populations, were found in close proximity to each other (Fig. 4.17, WAG 2014). The Alps and the Eastern Alps in particular can therefore be expected to become a melting pot of various European wolf populations, enhancing the genetic diversity of the overall Alpine population. This process has already been initiated by the mating of a male disperser (Slavc) from the Dinaric population with a female from the Italian population in Veneto (see Box 7.2. and Fig. 7.4 in Chapter 7.1.) This could in the long run mean that current subspecies such as *Canis lupus italicus* may lose their distinctness and therefore their status as subspecies.

One of the challenges regarding the conservation of the isolated Italian population in the past and present is the hybridization with dogs (Boitani 1983 in Boitani 2003). “The introgression of dog genes may reduce the viability of wolf populations with the destruction of adaptation, and provide an increase in aggressive behaviour and livestock depredations” (Randi 2011). This was expected to happen around the edges of the range: dispersers might not encounter another wolf in those areas, but at the same time there are more than one million free-ranging dogs estimated to be in Italy (Genovesi & Dupré 2000 in Verardi et al. 2006). Both possible pairings pose specific challenges to the process of hybridization. Female wolves have a single oestrus cycle per year. Male wolves show seasonal increases in sperm production, testosterone and testes size (Asa 1997 in Vila & Wayne 1999). In contrast, females in most dog breeds can produce two litters per year and male dogs maintain elevated testosterone level all year round (Asa 1997 in Vila & Wayne 1999). On the one hand, a male wolf and a female dog may not be well timed for interbreeding when they encounter each other. On the other hand, a male dog and a female wolf can potentially mate during peak receptivity. However, dispersing female wolves do not return to their natal pack, but start a new pack with their mate. Male dogs often do not form long-term bonds with females, or assist in the upbringing of offspring. Therefore, pup mortality is thought to be very high, like in feral dogs in Italy and surviving hybrids may have difficulty integrating into a wolf pack (Vila & Wayne 1999).

The genetic data show only few occurrences of wolf x dog hybridization. Boitani (1982) reported a known case of hybridization when a radio-tracked female wolf bred with a male feral dog and subsequently gave birth to six cubs. Neither Randi et al. (2000) nor Nowak & Federoff (2002) found evidence of hybridization in their data. Randi & Lucchini (2002) found only one of 107 individuals (0.9%) to be mixed. However, Verardi et al. (2006) found eleven of 220 individuals (5%) to be likely admixed. A similar result was found by Caniglia et al. (2014) who detected 16 hybrids (4%) in their samples from the northern Apennines. They also remark that many previous studies did not analyse enough genetic markers to have a high enough power of recognizing hybrids. In addition, studies analysing mtDNA can only identify hybrids between female dogs and male wolves, as mtDNA is inherited maternally (Randi & Lucchini 2002).

Hybridization of wolves and dogs does not only occur in the wild. There are dog breeds that are wolf x dog hybrids, e.g. the Saarlos wolfdog and the Czechoslovakian wolfdog. The breeding of the former started in the 1930s, and the last crossbreeding with a wolf occurred in 1963 (SWH-Club 2014). The breeding of the latter started in 1955 with the crossbreeding of a German shepherd dog with a wolf from the Carpathians. The last genetic enhancement by crossbreeding with a wolf happened in 1982 (TWH-Club 2014). Genetic analyses still recognise both breeds as hybrids, and samples of them are sometimes used as known hybrids for comparison of wild samples (Verardi et al. 2006). Both breeds still look wolf-like and when owners let them run around loose, they are sometimes reported as wolf sightings by members of the public (R. Manz, KORA, pers. comm.). Few samples collected in the field and identified as hybrids could indeed originate from domesticated hybrid dogs.

4.4. Discussion and conclusions

The reliability of any assessment of the conservation status of a species directly depends on the quality of the monitoring. Surveying large carnivores – as a matter of fact, any wildlife species – is a difficult and if properly done also a costly endeavour, but it is a legal requirement in all countries of the Alpine convention for game and for protected species. Expert groups – the SCALP group for the lynx and the WAG for the wolf – have, based on the experiences from the countries where the respective populations established first, recommended scientific robust procedures for monitoring lynx and wolf and for an Alps-wide cooperation that can be applied in each country, adapted to the regional requirements. But still, not all national or regional wildlife authorities in charge have implemented these protocols, and for many regions across the Alps, the number of lynx and wolves is rather a guess than a qualified estimation. All in all, resident large carnivores like wolf or lynx in a intensively observed region such as the Alps do not go undetected for a long time, and properly trained staff can confirm their presence. Transient dispersing animals are more difficult to catch, but both lynx and wolfs will eventually settle down (and become more visible), even if they are isolated.

We assume that “false negative records” (undetected large carnivores) create a much smaller bias than “false positive records”, as indirect signs (and even direct sightings) can easily be misinterpreted by the unexperienced observer and because the high media attention of large carnivores provokes such confusion. A strict classification of the observations, e.g. according to the so called SCALP categories, helps to increase the certainty. C1, C2 and C3 reports are in a certain balance in areas that are permanently settled, and if from a given area only C3 are reported, the species is either absent or there is something wrong with the monitoring system (e.g. lack of trained people to confirm occurrence).

Nevertheless, resident animals of both species, especially if they reproduce, are rather obvious and are very unlikely to go undetected in the Alps. Wherever we have populations (e.g. in the southwestern Alps of Italy and France for the wolf and in the Swiss Alps for the lynx), a reliable monitoring is established, although even there the challenge is to secure the long-term means to continue the monitoring. Wherever new population nuclei emerge, a reliable monitoring system needs to be established. Transient and isolated single individuals do not much contribute to the whole population (e.g. the over-all assessment does not significantly vary whether these animals are included or not), but to detect and observe these individuals is important to anticipate the population development and to prepare the monitoring, inform the local population, and eventually implement management measures such as livestock protection protocols.

Both the Alpine lynx and the Alpine wolf populations were assessed as Endangered (Chapters 4.2.2 and 4.3.2) and the threats listed (e.g. persecution, low acceptance and inbreeding for lynx; low acceptance, poaching and poor management structures for wolf) are indeed crucial. “Habitat loss due to infrastructure development” was additionally listed for both species by Boitani et al. (2015). This would be an important point, e.g. for lynx, if it leads to further fragmentation. For both species, the low acceptance especially by the land users in the areas where they so far have settled leads to conflicts, which are difficult to mitigate and often lead to illegal killing. Anthropogenic losses – even if they are in a range that would be sustainable for a “normal” population – can strongly affect small populations. We e.g. assume that the very slow growth of the lynx population (not more than 130–180 mature individuals in the whole of the Alps 40 years after the first reintroduction) was a consequence of the high losses due to illegal killings, and that this slow growth has (besides the too narrow founder group) substantially contributed to genetic drift and the resulting genetic depletion. Inbreeding is probably the most serious long-term threat to the survival of the remnant lynx population(s) in the Alps.

This is not the case for the wolf. Although the Italian population, the main sponsor of the emerging Alpine population, went through a historic bottleneck, it still has a sufficient genetic diversity, and as the Alps will be a melting pot of several European wolf populations, the future genetic diversity of the Alpine wolf population will be the highest in Europe. Indeed, although both species at present are assessed as Endangered in the Alps, the population dynamics of the two carnivores is totally different. Whereas the wolf shows increasing tendencies almost everywhere, the lynx is stagnant at best. This may be partly due to the different genetic constellations of the two populations (although there is no hard evidence for an *inbreeding depression* in lynx so far), but it is certainly a consequence of the differences regarding the socio-spatial and ecological traits of the two species, which will be addressed in the following chapters.

“Poor management structures” was another threat identified for lynx and wolves. The major challenge seems to be the transition from a “passive management” (or *laissez faire*, the species are strictly protected...) to an “active management”, which would require the definition of goals, a broad societal agreement, and active measures.

5. Ecological factors: People, habitat and prey

Homo sapiens is the most dominant species in the Alps, defining the ecological parameters for the existence of almost all other species. The Alps are the most intensively used mountain range in the world, and the human impact on the landscape and ecology of the Alps is vast. People have a direct – through anthropogenic mortalities – but even more an indirect effect on large carnivores, as they are also shaping their habitats and prey. However, although more people live in the Alps nowadays than in the 19th century, when the large mammals were dwindling, the impact of man on the environment has not linearly increased, but dramatically altered. Whereas 150 years ago the main impact came from the agricultural activities of the resident people and the over-exploitation of natural resources and habitats, todays main effects on the landscape of the Alps are the consequence of infrastructure construction and an intensive use of the landscape for recreation.

5.1. Development and distribution of human population

There are 5,867 municipalities in the Alpine Convention (PSAC 2010a) and they are home to about 15.2 million people (Bätzing 2015). Considering the Alpine area of about 191,000 km^2 this corresponds to a population density of almost 80 inhabitants per km^2. Additionally, about 60 million people visit the Alps for daytrips, and a further 60 million visitors stay for a total of 370 million nights in the Alps every year (Siegrist 1998). However, only an estimated 25% of the area is permanently inhabited, resulting in a very uneven distribution of the resident population in the Alps (Fig. 5.1; Tödter & Hasslacher 1998), and also tourism is spread unevenly over the seasons and the Alpine region. In 2011, 57% of the resident population was living in municipalities with their centre below 500 m although the area only covered 23% of the Alps. Only 5.6% of the population lived in the area above 1,000 m which makes up 26% of the Alps (Bätzing 2015). Looking at the historical population development helps understand the current situation.

5.1.1. Demographic development of the resident population

Since the eradication of the large carnivores from the Alps in the late 19th century (Chapter 3.1.1), the human population in the Alps has seen considerable changes in various forms.

Population development 1870–2011

Bätzing (2015) identified three phases: 1871–1951, 1951–1981 and 1981–2011. In the first phase, the population in the Alps increased from 7.8 million to 10.8 million. This increase (37%) was less than the average increase in the countries sharing the Alps (51%) and was caused by the industrial development of a few Alpine centres, better transport and the rise of the "Belle-Époque"-tourism. In the western part of the Eastern Alps (Bavaria, western Austria), almost all municipalities experienced a population increase. However, regions without these new economies, experienced a decrease of their population, e.g. the south-western Alps in France and Italy. The Alpine area was also heavily influenced by the two world wars within that time span, which stopped large parts of the economic dynamics.

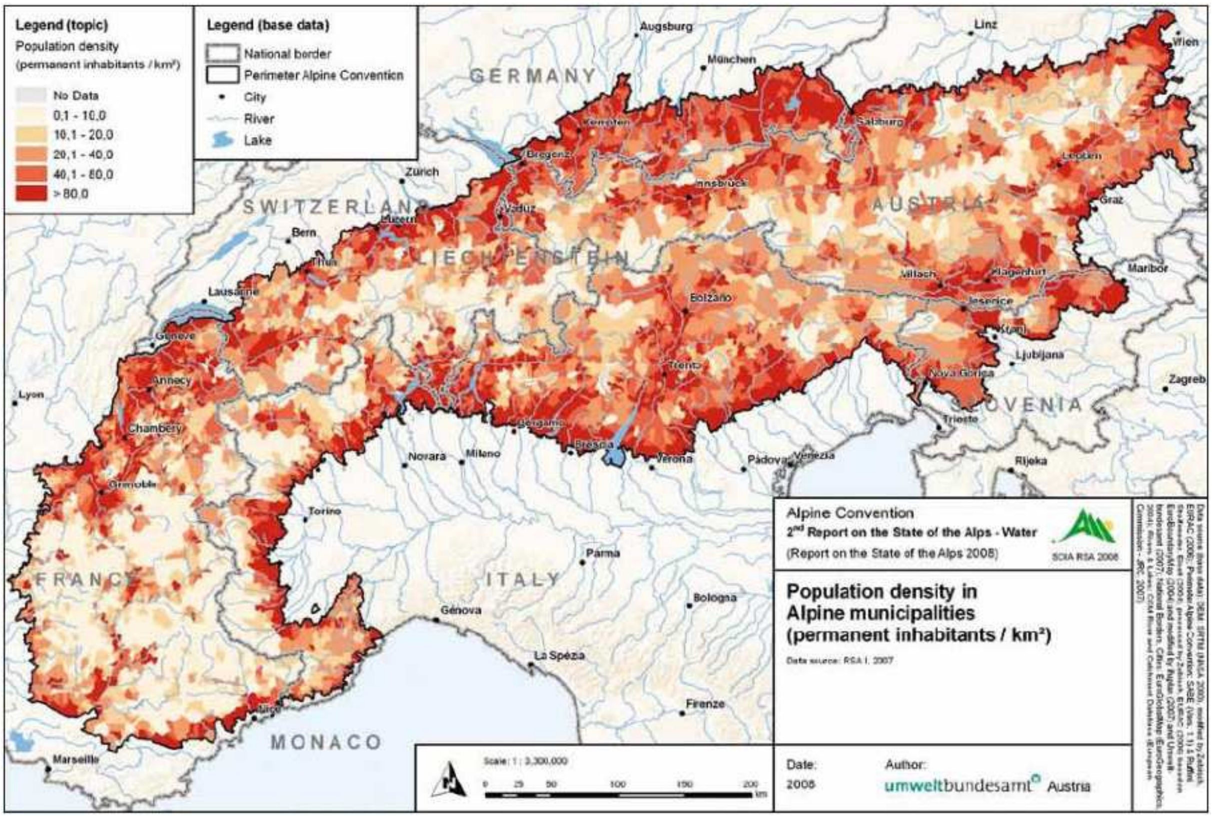


Fig. 5.1. Human population density in Alpine municipalities (PSAC 2010b).

The second phase from 1951–1981 was mainly influenced by a restructuring of the European post-war economy. The importance of north-south connections overtook the previously dominant west-east connections. This led to a dramatic increase in traffic through the Alps, resulting in a gain of importance of cities and towns along these north-south routes. This period was also the start of mass tourism, which created many additional infrastructures and labour, respectively, in the Alps. Additionally, new industries (e.g. hydro power plants) provided further job opportunities. Mechanisation of the agriculture was however limited because of the steep slopes, and Alpine farming became increasingly unprofitable.

The resident population increased from 10.8 to 13.1 million people during this phase, marking the strongest increase in the period between 1871 and 2011. However, municipalities, which were not able to profit from the new economies, saw their population decrease. While the French part of the south-western Alps started to profit from tourism and the population in about half of those municipalities increased again, the population of the regions of the southern and south-eastern Italian Alps (Veneto, Friuli, Lombardy) and eastern Austria started to decrease.

In the third phase (1981–2011) the total population increased from 13.1 to 15.2 million people. The population growth rate however decreased, though less prominent than the average decrease in Europe. On the one hand, the Alps profited from the new European phenomenon of “decentralisation” and the higher mobility. The population of settlements along important routes continued to increase. Some cities at the edges of the Alps started to become “commuter towns” for larger cities outside the Alps (e.g. Lyon, Milan, Vienna, Munich). On the other hand, tourism started to stagnate in the mid-eighties and the population of small tourism regions tended to decrease. The industrial boom came towards an end, leading to a shut-down of many branches and loss of job opportunities.

The population decrease continued in the Italian Alps and in eastern Austria (Fig. 5.2), resulting in population densities of less than 20 inhabitants per km^2.

Over the whole period from 1871 to 2011, 59% of all Alpine municipalities, representing 60% of the Alpine area, experienced an increase of their population by a factor of 2.5 on average. The remaining 41% of the municipalities lost on average 40% of their inhabitants. The distribution of the population across the Alps is nowadays a lot more uneven than in 1870. The observed population growth was driven by municipalities with their centres below 1,000 m.

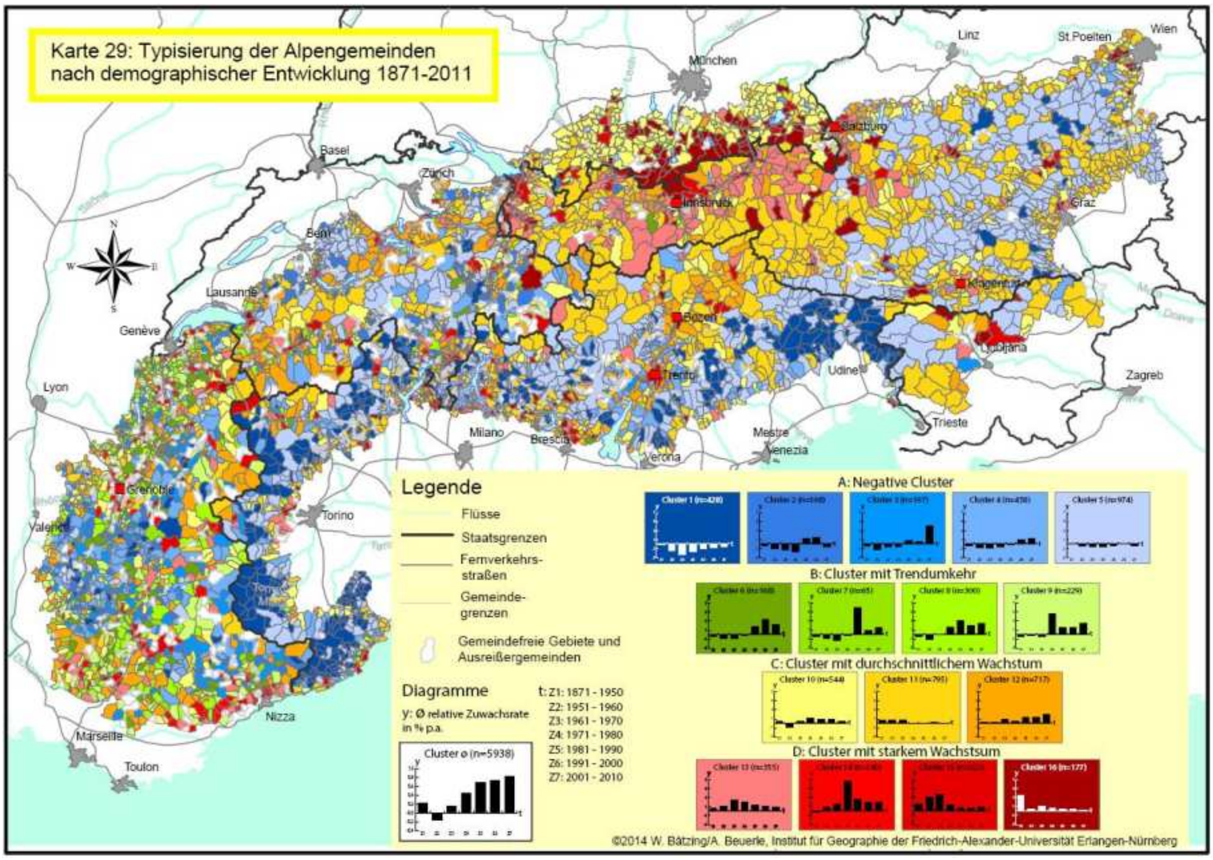


Fig. 5.2. Summary of population development in the Alps 1871–2011 by municipality (Bätzing 2015). Blue: overall population decrease. Population in the blue clusters 3 & 4 started to increase again (see diagrams in “Legende”) but are still lower than in 1871. Green: increasing population after an original population decrease. Mostly tourism destinations or “commuter towns” (see text). Yellow/Orange: Stable population growth with average growth rate. Red: above-average growth, mostly urbanised areas.

Current situation and challenges

52% of the municipalities are considered “rural”, however, 75% of the population live in urban areas (Bätzing 2015). The biggest urban area within the Alps is Grenoble with 664,832 inhabitants in 2008 (urban core: 495,429; periphery: 169,403; Insee Rhône-Alpes 2011), but a third of all Alpine municipalities have less than 500 inhabitants (Table 5.1; PSAC 2007).

Although the total number of jobs in the Alps increased between 1981 and 2001 from 2.7 million to 3.1 million (without Italy and Slovenia), job opportunities decreased in 40.5% of all municipalities (Pfefferkorn & Musovic-Doboš 2007). In 2001, the commuter balance for the entire Alpine region was minus 487,217; i.e. the number of people living within the Alps but working outside is almost half a million higher than the number of people travelling into the Alpine region to work. Urban and tourism municipalities show a positive commuter balance, while suburban and peripheral municipalities contribute the majority to the negative commuter balance (Pfefferkorn & Musovic-Doboš 2007). All the commuters working in the peripheral metropolises are happy to live in a place away from the city with recreational opportunities, but the traditional Alpine economies or the sustainable (economic) regional development are of little interest to them (Bätzing 2013).

Table 5.1. Municipality types and population structure in the Alps (incl. Monaco; PSAC 2007). Date of survey: AT: 2005, DE, IT, LI, SL and CH: 2004, MC: 2000, FR: 1999.

Population classes	Number of municipalities	Share on total number of municipalities [%]	Number of inhabitants	Share on total population [%]
<500	1,876	31.5	445,588	3.2
500 - <1,000	1,099	18.5	797,585	5.7
1,000 - <2,500	1,572	26.4	2,551,301	18.2
2,500 - <5,000	816	13.7	2,810,900	20.1
5,000 - <10,000	367	6.2	2,476,149	17.7
10,000 - <25,000	175	2.9	2,522,397	18.0
25,000 - <50,000	35	0.6	1,166,367	8.3
≥50,000	14	0.2	1,228,738	8.8
Total Alps	**5,954**	**100.0**	**13,989,025**	**100.0**

Outlook

Areas with a decreasing population consequently experienced infrastructural problems. Schools, health care centres, shops and restaurants are closed, and more people move away, especially younger ones and families (Bätzing 2015). A population is considered as “over-aged” if over 15% of the population is older than 60 years (PSAC 2007). In 2000, according to this definition the population in almost two thirds (63%) of Alpine municipalities was over-aged. About 41% of these were located in Italy, and more than a quarter in France (PSAC 2007). Another measure for over-ageing is the Old Age Index, which is calculated as the number of people over 64 per 100 people under 15 (PSAC 2007). The Alpine average for the Old Age Index is 100.3, reaching from 241 (Liguria) to 63 (Liechtenstein) and 64 (Vorarlberg; Fig. 5.3; PSAC 2007).

In the regions with an over-aged population, a further population decrease can be expected. Bätzing (2013) presents a scenario for the Alps until 2033 based on the continuation of current developments: The economies will belong to companies outside the Alps (energy and tourism), or will have disappeared (agriculture and artisanry). The population in mountain areas will leave and the peripheral Alpine areas near the metropolises outside the Alps will be further urbanised. The Alpine region disaggregates into the urban catchments of the eleven metropolises surrounding the Alps[5], who dominate their respective area. Large areas of ‘no man’s land’ lie in between, with a few tourism centres (Bätzing 2013). The same view is shared by Pfefferkorn et al. (2005) as reproduced by CIPRA (2008) who add that this will also be “*characterised by intensification of land use on the one hand and abandonment and reforestation on the other hand*”.

[5] Berne, Geneva, Genoa, Ljubljana, Lyon, Marseille-Nice, Milan, Munich, Turin, Vienna and Zurich.

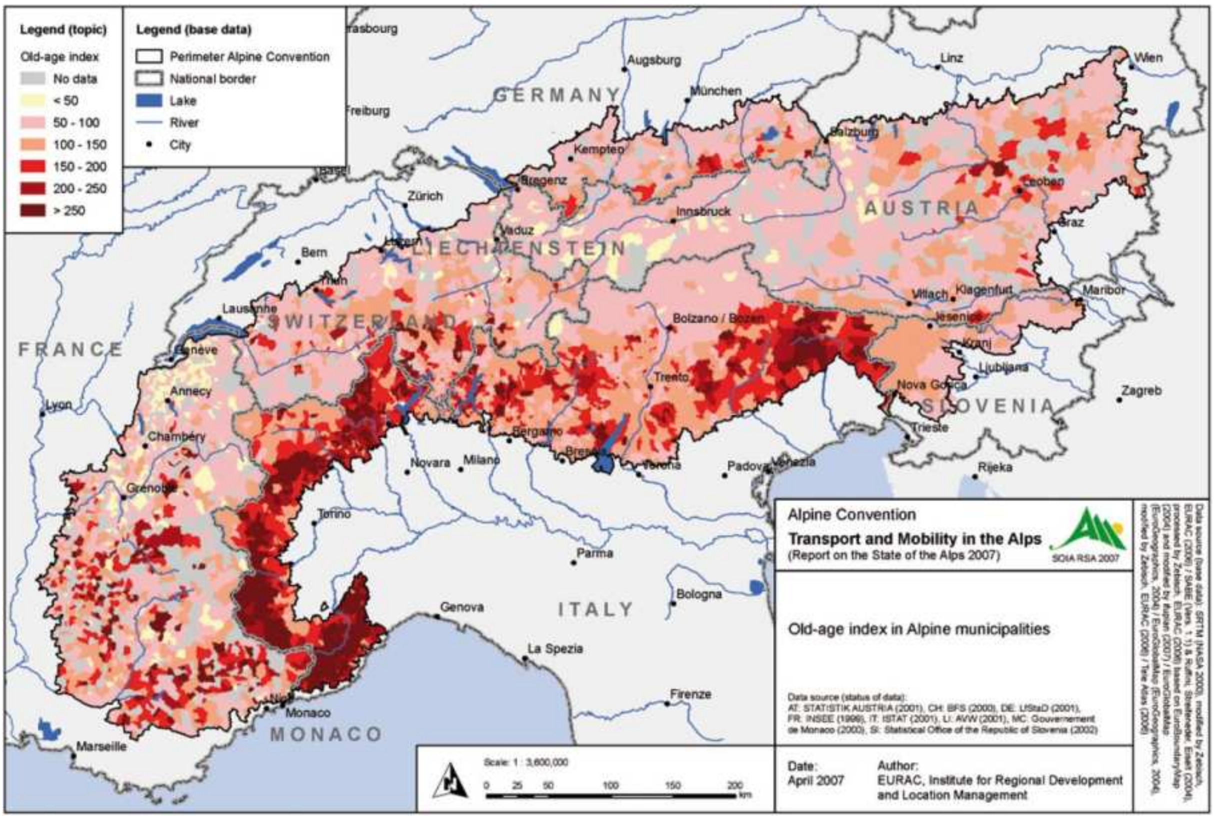


Fig. 5.3. Old-Age Index (number of people over 64 years of age per 100 people under 15) in Alpine municipalities (PSAC 2007).

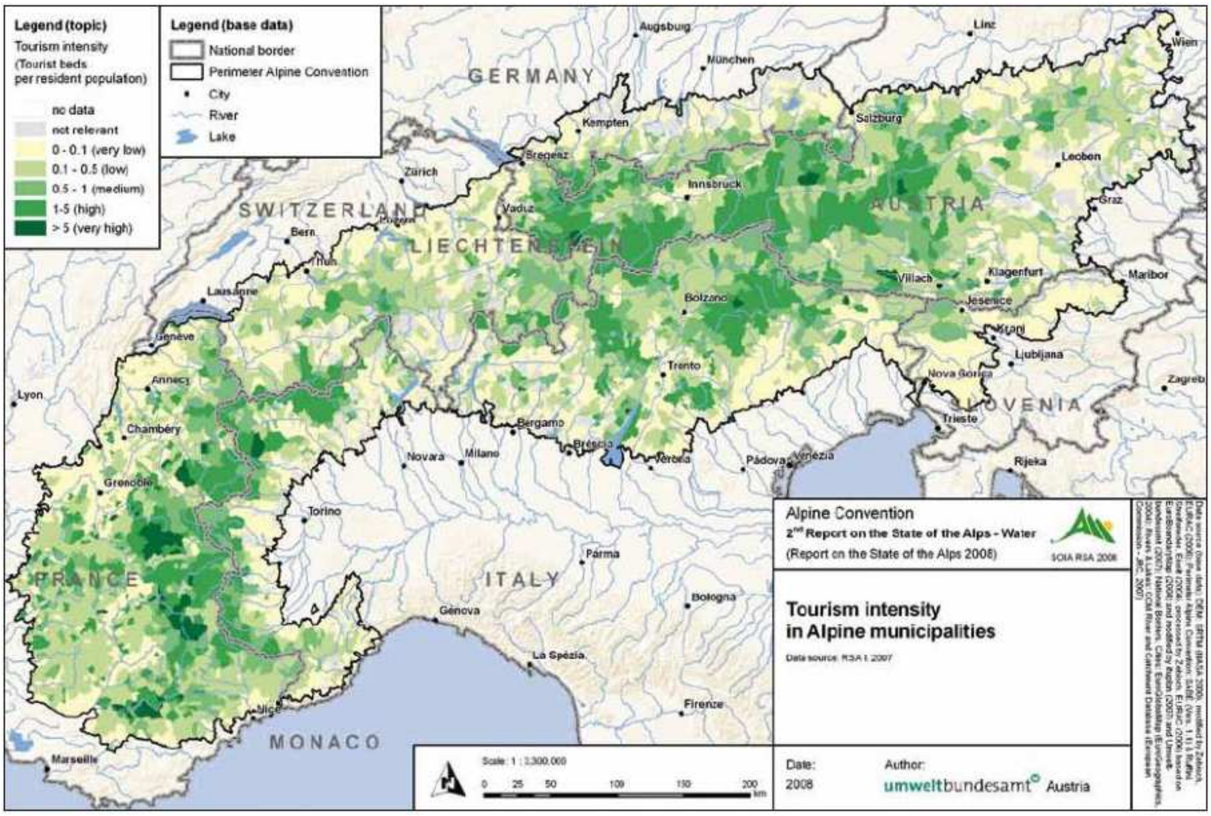


Fig. 5.4. Tourism intensity (tourist beds per resident population) in Alpine municipalities (PSAC 2010b).

5.1.2. Tourism (non-resident population)

Tourism was booming in the Alps between 1955 and 1982 (Bätzing 1997). While the number of overnight stays has remained stable since then, the number of tourist beds decreased slightly (Bätzing 2013). International arrivals in the Alps amounted to over 33 million in 2010 (BAKBASEL 2011[6]). An estimated 60 million people were visiting the Alps every year for daytrips in the mid-1990s. The total number of arrivals amounted to a further 60 million per year, staying for about 370 million nights (Siegrist 1998). 15% of all jobs in the Alpine region are in the tourism industry (BAKBASEL 2011, Bätzing 2013).

Some 4.5 million beds were available to tourists of which about 1.2 million were provided by hotels (Bartaletti 2008). However, in 1995 the capacity utilisation amounted to only 22% (Siegrist 1998). On average about 1 million non-residents are staying in the Alps every night. However, just as the permanent population, tourism is spread unevenly across the Alpine region and seasons. More visitors arrive in summer than in winter, and 46% of beds are offered by 5% of the municipalities, while 37% of Alpine municipalities offer no tourist beds at all (Price et al. 2011). In 8% of Alpine municipalities, the number of available tourist beds is higher than the number of inhabitants (Fig. 5.4; Bätzing 2003).

Ecological effects of tourism

Tourism can have direct and indirect ecological impacts. Among the direct consequences are the reshaping of areas (e.g. transport infrastructure, ski slopes, or golf courses), and effects from the physical and chemical properties of artificial snow on ski slopes. Indirect effects include the urbanisation of settlements and the increase of the population, splinter development and urban sprawl (towns relying on tourism need the same space as a non-tourism municipality with an estimated three to five times more inhabitants), and noise and air pollution through the increase in traffic. The indirect impacts are considered to be much worse than the direct impacts for the Alpine ecology as a whole (Bätzing 2003). In tourism-dominated municipalities, economy rules over social, cultural and environmental issues (Hauser 2007).

Tourism & large mammals

Large carnivores can be very attractive to tourism (e.g. lions in Africa, tigers in Asia, grizzly bears in North America). For some visitors, large carnivores represent pure nature and untamed wilderness. This image can be even more important than the actual presence of those animals themselves (Goodwin et al. 2000). There are already some locations in Europe advertising the presence of large carnivores, e.g. the Mercantour National Park in the French Alps (Goodwin et al. 2000). However, not only large carnivores can be used to attract visitors, but (large) wildlife in general: the red deer rutting period is presented as one of the main attractions of the Swiss National Park (Swiss National Park 2014a). A well-managed tourism based on wildlife (observation) can have several effects: the area can offer added value for holidays; tourism can increase the local community's sense of involvement in protecting their wildlife; local businesses can benefit from tourism; the visitors may be educated about the challenges in conservation; and tourists can become actively involved in research projects (e.g. as volunteer wolf trackers in the Carpathian Mountains, Romania, Goodwin et al. 2000).

The examples presented by Goodwin et al. (2000) however came all from National Park areas. The lynx trail near Lenk, Switzerland (Lenk Bergbahnen 2014) is a rare example of initiatives outside pro-

[6] The definition of the Alpine area by BAKBASEL is based on the Alpine Convention, but differs from it. For example, it includes the whole of Slovenia but no Italian areas bordering France apart from the Aosta Valley (see Fig. 3-1 in BAKBASEL 2011).

tected areas. In another case, the reintroduction of lynx in north-eastern Switzerland led to inquiries about the touristic opportunities based on the new lynx occurrence (Robin & Nigg 2002). However, this case also demonstrated the problematic aspects of such plans. While conservation groups, teachers and media expressed a positive opinion, land users tended to show a neutral or even negative attitude to the idea. Hunters argued against the plan, fearing additional tourists would have a negative impact on game (Robin & Nigg 2002). All in all, the typical mass tourism of the Alps makes very little use of wildlife as an attractant for visitors. Most visitors are unaware of the conservation needs of the wildlife sharing their recreational areas.

Wild animals show a variety of responses to disturbances from recreational activities (e.g. hiking, skiing, mountain biking, paragliding). Reactions are generally fleeing and avoidance of certain areas (Ingold 2005). This can result in the separation of mothers and young, increased energy costs and reduced physical condition, reduced access to favourable areas caused by disturbances, and eventually in decreased survival or reproduction (Ingold 2005). The distribution and density of infrastructure (e.g. roads, hiking paths, ski areas), and the frequency of their use are also relevant for wildlife conservation, as it may lead to increased habitat fragmentation (Chapter 5.2) with remaining suitable areas too small for large mammals to linger or for birds to breed (Ingold 2005). On the other hand, large mammals have the potential to adapt to the presence of people and to human activities. Despite the increase of tourism and recreational activities during the past 60 years, populations of ungulates in the Alps also increased (Chapter 5.3). Large carnivores with their adaptable behaviour and capacity to learn in particular have a high potential to live also in human dominated landscape and to be tolerant towards human activities (Chapter 7.2, 7.3).

5.2. Development and fragmentation of suitable habitats (forest)

Although lynx and wolves are in their entire distribution range not restricted to forested areas, the landscapes so far re-occupied in Central and Western Europe are characterised by a high share of forests (Chapter 7.1) and all habitat models for the Alps reveal a high affinity to forests (Chapters 7.2, 7.3). Consequently, our main focus for this chapter is on the development and fragmentation of forest habitat in the Alps. Forests have considerably changed since the all-time low of the large carnivores and their prey at the end of the 19th century. Generally, they have expanded and grown from over-exploited and heavily grazed woods into heavily managed high stands.

5.2.1. Development of forest area in the Alps

Forests have been massively overexploited and their area has decreased for centuries in Europe after man's change from hunting and gathering to arable farming and animal husbandry. The natural timberline in the Alps has been lowered in some places by almost 400 metres to expand the area of available summer pastures (Burga & Perret 1998). In Germany, the area covered by forest decreased between 650 AD and mid-14th century from 90% to 15% (Bork et al. 1998). In general, the reduction of forest area occurred later in the mountains than in the lowlands, but during the early stage of the industrialisation, even the least accessible forests in the Alps were exploited. The lowest extent of forests in the Alps was reached towards the end of the 19th century. Since then, the forested areas have expanded again (Fig. 5.5; PSAC 2011).

The forested area in the Alps has doubled between 1900 and 2012 (Bätzing 2015), as a consequence of protection of remnant forests and calculated afforestation. In the western Alps, the main motivation was protection from natural disaster (flooding, landslide, avalanches), in the eastern Alps the driving force was the increasing economic value of forests (e.g. high demand for timber and wood charcoal by the industry), and in the southern Alps, forests naturally started to recover as a consequence of rural exodus (Breitenmoser & Breitenmoser-Würsten 2008). The highest increase occurred in the most eastern parts of the Alps (Lower Austria, Upper Austria, Styria and eastern Carinthia) where in the second half of the 19th century wealthy entrepreneurs started to buy and afforest land also for hunting. Nowadays, there are approximately 120 contiguous municipalities with forest coverage of more than 80% in this area (Bätzing 2015). Forests have not only spread but their structure has also changed. Besides the expansion of forested areas, the quality of forest has considerably changed over the past 100 years: The forests became denser and the growing stock increased (Stöhr 2009), with a marked shift from coppice and "Mittelwald" (semi-open, grazed woods) to high forest (Bürgi 1999).

The CORINE Land Cover (CLC) survey of 2006 showed that 52.3% of the Alps are covered with forests and transitional woodland shrub, followed by pastures and mosaic farmland (14.5%), and open space with little or no vegetation (11.8%; EEA 2010, data without Switzerland). The three CLC surveys of 1990, 2000 and 2006 showed less than 1% of changes in land cover between the surveys[7]. The majority of the changes consisted of "forest creation and management" and contributed to 64.94% of land cover changes in the Alps between 1990 and 2000, and 58.70% between 2000 and 2006 (EEA 2010).

Passive and active reforestation was a very prominent, but not well-documented process; it is hard to find data on the long-term development. The definitions of "forest", the measuring techniques and the reference areas have changed so often that it is difficult to compare even small areas (Breitenmoser & Breitenmoser-Würsten 2008).The development of forested areas in the northern and eastern Alps are probably well-illustrated by the time series shown in Fig. 5.5, although they are not fully consistent with the geographic outlines of the Alps.

The Alpine forests are of high economic value: the export of wood products from Austria generates 8.5 billion € in annual revenues (Stöhr 2009), while the annual value of Swiss forests for recreation has been estimated at 10.5 billion CHF (Ott & Baur 2005).

[7] The „Arealstatistik der Schweiz" showed a slightly higher percentage of changes in the Swiss Alpine area for the time period 1985-2009. The changes also mostly came from an increase in forested area, at the expense of agricultural land (Bätzing 2015).

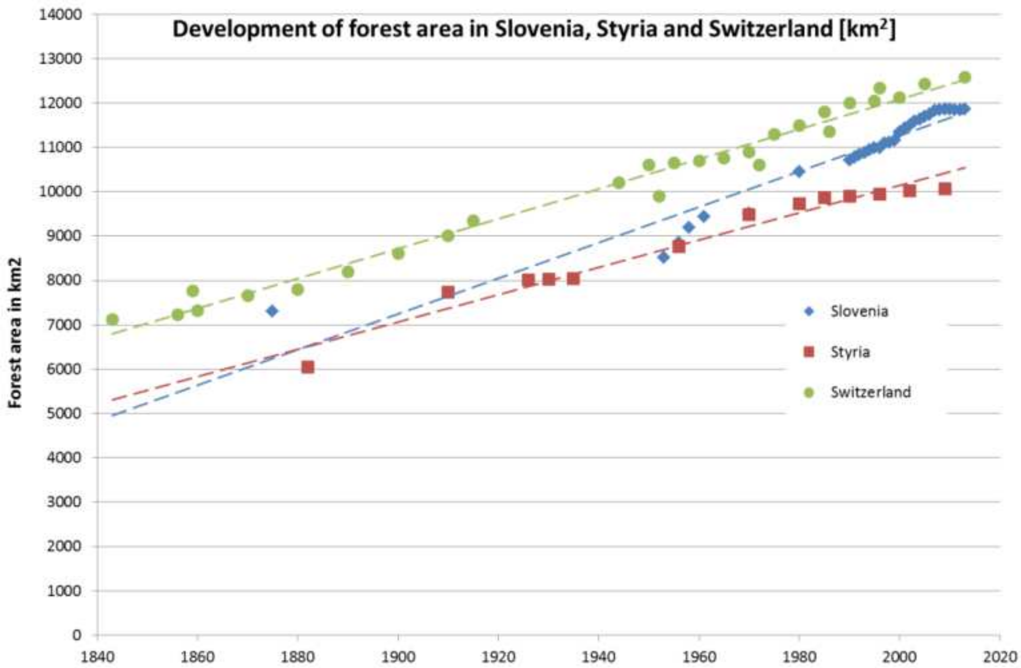


Fig. 5.5. Development of the forest area in Slovenia, Styria and Switzerland in km^2. Dashed lines represent linear regressions. Data for Switzerland and Slovenia include areas outside the Alps (Sources: Slovenia: Slovenia Forest Service 2014; Styria: for original sources see Breitenmoser & Breitenmoser-Würsten 2008, complemented with data for 2009 from Bundesforschungszentrum für Wald 2014; Switzerland: for original sources see Brändli 2000, complemented with data for 2005 and 2013 from Bundesamt für Statistik 2014).

5.2.2. Fragmentation of forest area in the Alps

Besides the extent and quality of forests (which have improved over the past 100 years in the Alps), the connectivity between forest patches is decisive for far-roaming terrestrial species. Habitat loss and fragmentation are the leading human-caused deterministic factors affecting wildlife populations with effects being caused by e.g. altered connectivity or increased edge effects (Mills 2007). Fragmentation was suggested as an indicator for monitoring sustainability of human land use (Moser et al. 2007). "*Over time, the process of forest (de)fragmentation relates to three main alterations in the landscape mosaic: [...] insufficient (sufficient) total forest habitat area, isolation (connectivity) of forest habitat patches, and shift of land uses at edges where forest habitat areas abut modified ecosystems (interface zones)*" (Estreguil et al. 2012). Fragmentation has effects on the environment and various ecosystem services. Animals are susceptible to traffic collisions with the increasing road and rail network. Indirectly, roads also lead to reduced prey availability due to increased access for hunters and poachers (Zimmermann 2004). Fragmentation leads to higher levels of disturbance and stress, loss of refuges, reduction or loss of habitat, barrier effect, disruption of seasonal migration pathways, genetic isolation, and reduction of habitat below required minimal areas and loss of species (Jaeger et al. 2011).

Various indicators have been used to measure forest fragmentation in the Alps and Alpine countries. The results of these studies highly depend on the definition of barriers – the "fragmentation geometry". Therefore, these definitions are included in the following descriptions. Generally, the degree of fragmentation is over-estimated towards the boundaries of the study areas or administrative units, as the limits of the observed areas are often treated like barriers, even though there may be no natural or anthropogenic barrier delimiting the studied areas.

Average size of remaining non-fragmented land parcels (average patch size)

Fragmentation of land and forests was analysed as an indicator for land use by the European Environment Agency (EEA 2003). Their analyses included the calculation per country of the average patch size and the frequency distribution of patches by size for non-fragmented land parcels. The same analysis was done specifically for forested areas as well (Fig. 5.6).

The loss of small patches is a weakness of this indicator. When small patches are lost, e.g. by urban sprawl, the average size of the remaining patches increases and suggests a positive development (Jaeger et al. 2011).

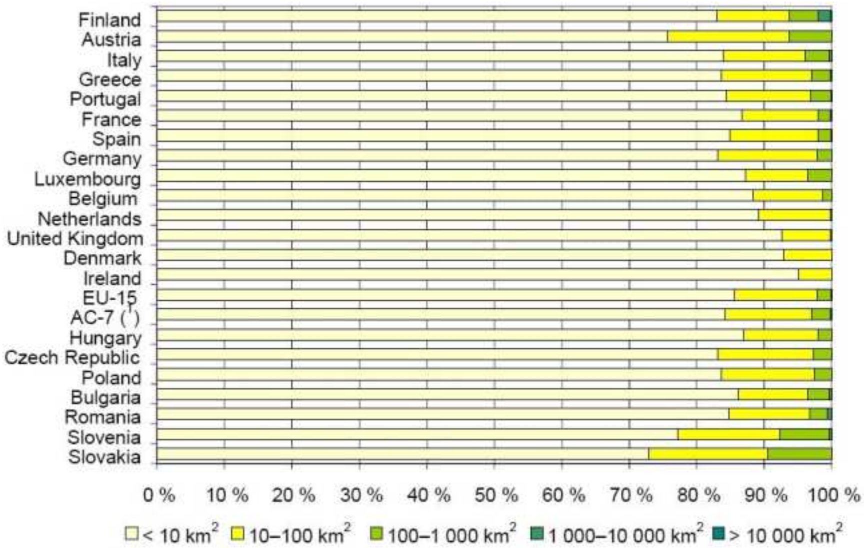


Fig. 5.6. Frequency distribution of forest patches by size (EEA 2003) in EU countries. NB: Data on forest patches refer to land cover data from 1986–97 for the EU and 1989–92 for ACs.
([1]) AC-7 refers to the ACs shown in the graph.

Number of remaining large unfragmented low-traffic forest areas above a certain size

Esswein & Schwarz-von Raumer (2006) analysed the remnant not fragmented forest patches for Bavaria. They compared three different fragmentation geometries FG. FG-1 included all roads of a certain class or higher, multi-tracked railways lines, settlements and canals. FG-2 also included settlements, but only roads with a traffic volume of more than 1,000 cars per day, railway lines of considerable importance, canals classified as national waterways of a certain category or higher, and airports. Tunnels longer than 1,000 m were regarded as a discontinuation of the fragmentation. FG-3 used the same elements as FG-2 but added a disturbance buffer of 1,000 m width around roads with a traffic volume of more than 10,000 cars per day (which are seen as insurmountable barrier for almost all animal species) and a buffer of 300 m width around roads with a traffic volume of more than 5,000 cars per day and multi-tracked railway lines. The Alpine region contributes a significant part to the number of these large unfragmented areas (UFAs). For FG-1, 8 out of 10 UFAs larger than 200 km^2 are found along the Alps and 18 out of 27 UFAs larger than 100 km^2 are also found there (Fig. 5.7). The results for the Bavarian Alps are also similar for the other fragmentation geometries. The increased number of large unfragmented areas for FG-2 and FG-3 mainly comes from other parts of Bavaria (Esswein & Schwarz-von Raumer 2006). For comparison, 400 km^2 are supposed to be large enough to sustain a lynx sub-population if it provides the needed resources (Becker 2013; Chapter 7.2.2).

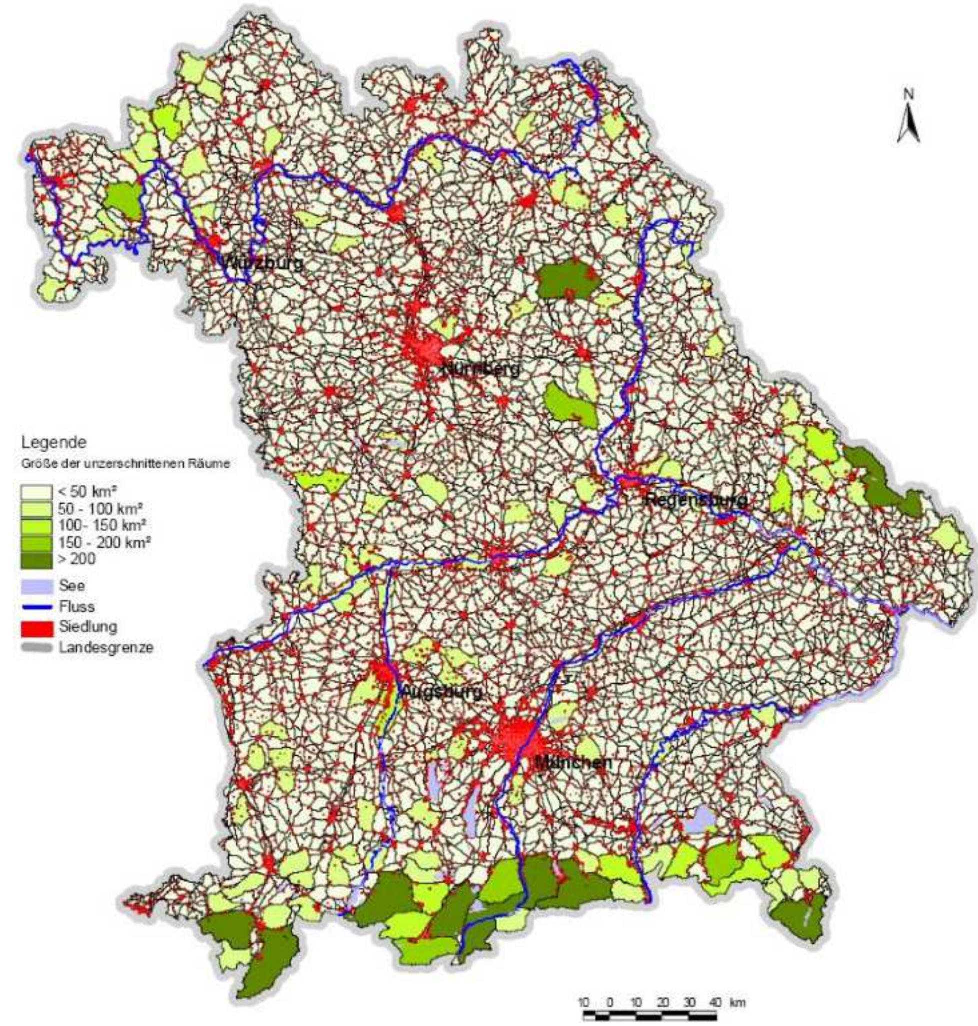


Fig. 5.7. Map of large unfragmented areas in Bavaria (Esswein & Schwarz-von Raumer 2006).

Table 5.2. Results of the IÖR-Monitor 2008 (Germany: 2010) and 2012 regarding large unfragmented areas in the Alpine administrative districts of Bavaria (IÖR 2014).

Administrative district	**% forested areas >50 km²**		**% areas >50 km²**		**% areas >100 km²**	
	2008	2012	2008	2012	2008	2012
Bad Tölz-Wolfratshausen	35.5	35.4	54.5	54.4	45.8	45.8
Berchtesgadener Land	17.3	17.5	61.1	61.1	40.4	40.3
Garmisch-Partenkirchen	41.4	40.6	87.2	87.2	79.2	79.2
Kaufbeuren (city)	0.0	0.0	0.0	0.0	0.0	0.0
Kempten (city)	0.0	0.0	0.0	0.8	0.0	0.0
Lindau	0.0	0.0	4.9	4.8	0.0	0.0
Miesbach	37.3	37.2	50.0	49.9	48.2	48.0
Oberallgäu	4.9	4.9	62.8	61.1	33.8	33.8
Ostallgäu	8.0	8.0	29.1	27.2	10.6	10.6
Rosenheim	12.6	12.5	23.8	23.8	10.7	10.7
Rosenheim (city)	0.0	0.0	0.0	0.0	0.0	0.0
Traunstein	15.2	15.2	29.8	29.8	29.8	29.8
Weilheim-Schongau	3.3	3.3	29.0	28.9	6.6	6.6
Germany (comparison)	3.6	3.5	17.4	17.0	6.7	6.5

The number of large unfragmented areas is an indicator of the monitoring of settlement and open space development ("Monitor der Siedlungs- und Freiraumentwicklung", IÖR-Monitor) by the Leibniz Institute of Ecological Urban and Regional Development (IÖR 2014). Additionally, the more specific large unfragmented forested areas are also an indicator. The monitoring covers the whole of Germany and the results for the Alpine administrative districts of Bavaria are presented in Table 5.2. The fragmentation geometry consists of settlements, and roads and railways above local level.

The results of Jaeger et al. (2007) were mainly presented for their FG-4. It included anthropogenic barriers, but excluded rivers, lakes and mountains higher than 2,100 m (i.e. natural barriers) from the reference area. Results with other FGs had shown that these natural barriers completely dominated the results and made a comparison of mountain and lowland areas almost impossible. Under FG-4, 26% of the area of Switzerland consists of UFAs larger than 100 km^2. They are almost exclusively found in the Alpine foothills and the Alps (Fig. 5.8). If lakes and areas above 2,100 m are included, the proportion of UFAs larger than 100 km^2 in Switzerland increases to 53% (Jaeger et al. 2007).

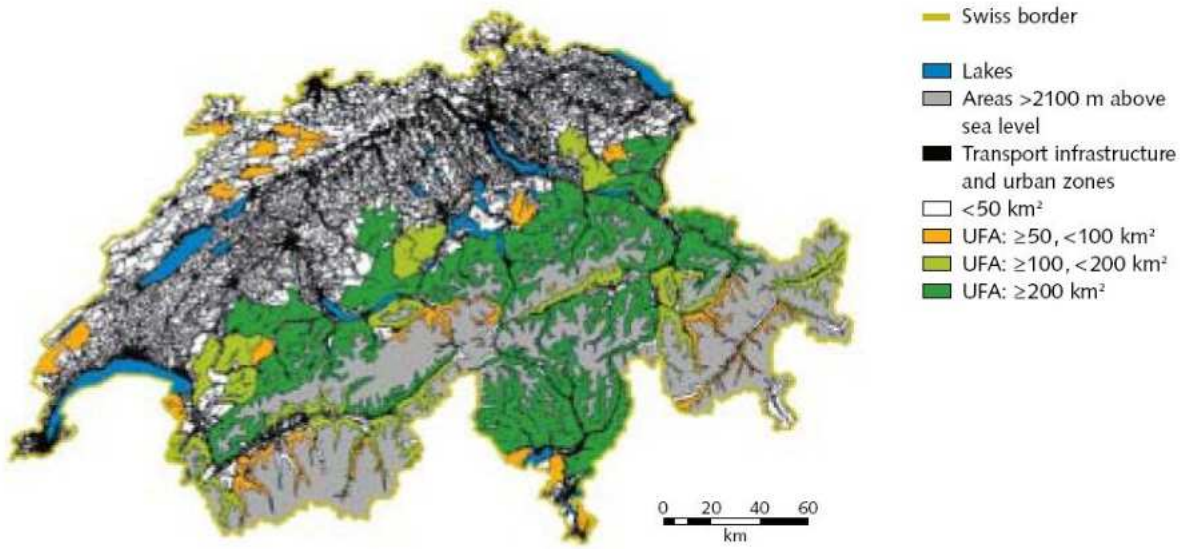


Fig. 5.8. Map of large unfragmented areas (UFAs) of 50, 100 and 200 km^2 in 2002 in Switzerland (Jaeger et al. 2007).

Using UFAs has its drawbacks. For example, a study regarding UFAs larger than 100 km^2 does not register changes within any of the unfragmented areas smaller than 100 km^2. (This may be not so important for resident large carnivores as a part of a population, but possibly for transient animals and hence for the connectivity between subpopulations.) If only the number of UFAs larger than 100 km^2 is given, some changes within that area are not registered either. For example, a decrease of the unfragmented area from 150 km^2 to 110 km^2 would not be registered. Moreover, if only the number of UFAs larger than 100 km^2 are considered, the division of a 300 km^2 area into two separate 150 km^2 areas results in an increase of the number of UFAs, thus wrongfully suggesting a positive development. Some of these weaknesses can be countered by not only giving the number of areas but also their size covered (Esswein et al. 2003).

Effective mesh size m_{eff} and effective mesh density s_{eff}

The calculation of the effective mesh size m_{eff} is based on the probability that two animals can find each other in the landscape, i.e. are located in the same patch (Jaeger 2000). This probability is then multiplied by the total area, converting the number into a measure of area – the mesh size of a regular grid pattern showing an equal degree of fragmentation. The effective mesh density s_{eff} is the effective number of meshes per area, i.e. the inverse of the effective mesh size $1/m_{eff}$ (Jaeger et al. 2011). An improvement of the methodology also avoids the problem of over-estimation of the fragmentation towards the (artificial) boundary of the study area (if the data are available) and for reporting the results for the administrative units within the study area (Moser et al. 2007).

Esswein & Schwarz-von Raumer (2006) calculated not only UFAs but also the effective mesh size in Bavaria for their three fragmentation geometries (see above). The resulting effective mesh size amounted to approximately 35 km^2 for FG-1, 65 km^2 for FG2 and 55 km^2 for FG-3. Within the 96 natural landscape units of Bavaria, they found the highest values of up to more than 300 km^2 in the Alpine regions for all three FGs (Esswein & Schwarz-von Raumer 2006).

Jaeger et al. (2007) have used the effective mesh size and density to analyse the development of landscape fragmentation in Switzerland between 1885 and 2002. FG-4 (where natural barriers were excluded from the reference area; see above) showed the highest increase in mesh density between 1960 and 1980 when the construction of many motorways was completed. The effective mesh size decreased in Switzerland from 580 km^2 in 1885 to 332 km^2 in 1935 and then to 176 km^2 in 2002. The mesh size in the Alps is higher than in the Swiss Plateau (10 km^2) and in the Jura mountains (20 km^2) and constitutes about 250 km^2 in the Central Alps and about 375 km^2 in the Northern and Southern slopes of the Alps. However, some valley bottoms within the Alpine eco-regions are as heavily, or even more heavily, fragmented as the Central Lowlands. The actual fragmentation has probably increased even more than the numbers suggest. For example, traffic volume has increased and roads today are wider (Jaeger et al. 2007).

The 116 municipalities of South Tyrol were analysed by Moser et al. (2007). The fragmentation geometry consisted of the road and railway network and the areas of development. The effective mesh size in South Tyrol was 495 km^2, ranging from 2.1 km^2 to 1,065 km^2. Municipalities in the sparsely populated mountain areas in the northeast and the west exhibited a high mesh size; municipalities in the central valleys with moderate population densities but major transportation axes revealed moderate mesh size; and the densely populated lowland areas in the South a small mesh size (Fig. 5.9).

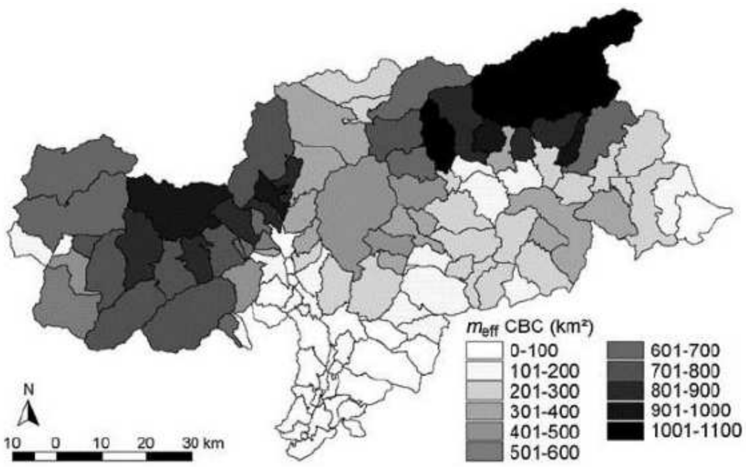


Fig. 5.9. Geographic distribution of the effective mesh size in the 116 municipalities of South Tyrol (Moser et al. 2007).

Jaeger et al. (2011) performed an analysis of the effective mesh size and effective mesh density for the whole of Europe. Their results are mainly reported for their fragmentation geometry FG-B2 which included motorways, roads up to a certain category, railway lines and built-up areas, and excluded natural barriers such as mountains, lakes and major rivers from the reference area. As in Jaeger et al. (2007), the removal of these dominant features enables a comparison e.g. of the Alps with regions without mountains or lakes. The results for the mesh density around the Alps per 1 km^2 grid are shown in Fig. 5.10. It is visible that the foothills (the lower areas) of the Alps have a low fragmentation, but that the valleys inside the Alps are rather more fragmented.

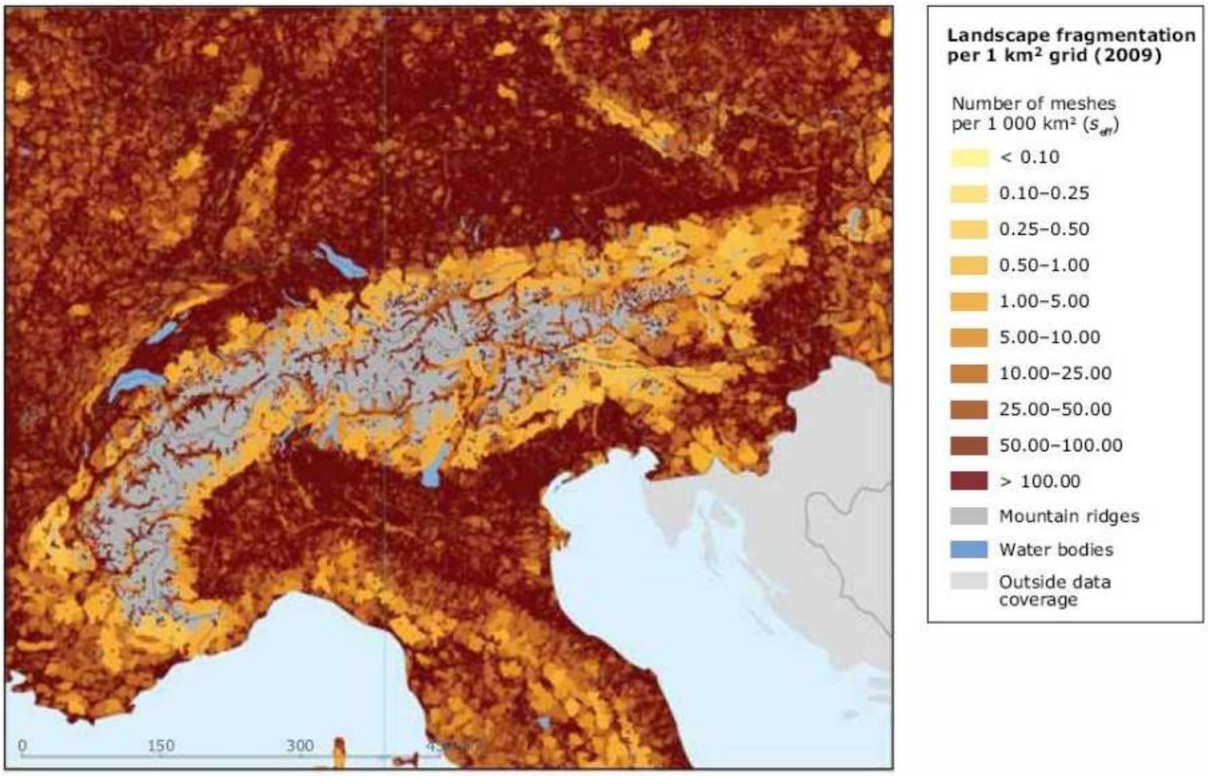


Fig. 5.10. Effective mesh density per 1 km^2 grid in the Alps and their vicinity in 2009. Natural barriers such as lakes, rivers and high mountains, which otherwise dominate the results, had been excluded from the reference area to enable comparison with areas without these features (Jaeger et al. 2011). The foothills of the Alps provide – compared to areas outside the Alps including secondary mountain ranges – still a large contiguous area with a rather low degree of fragmentation. However, valleys exhibit considerable fragmentation and can present barriers.

Continuum Suitability Index CSI

Under the Alpine Space Programme of the EU the ECONNECT project was created and ran from 2008-2011. The goal of the project was *"to improve the understanding of the concept of ecological connectivity and to enhance such connectivity across the Alpine range"* (Füreder et al. 2011). The web tool Joint Ecological Continuum Analysing and Mapping Initiative JECAMI[8] was a result of this project. To visualise the results of the project it used the Continuum Suitability Index CSI. The CSI was created to compare and visualize different indicators more easily. The index ranges from 1 = unsuitable to 100 = highly suitable as an ecological continuum. Concerning the fragmentation in the seven pilot regions, which were analysed in more detail, first the effective mesh size was calculated, before the results

[8] http://www.jecami.eu/ (last accessed on 25.11.2014)

were classified into the CSI values according to the scale given in Table 5.3. Unfortunately, the fragmentation geometry is not given in the layer description (Swiss National Park 2014b), hence it is not clear what kind of barriers were considered and how relevant they might be for lynx or wolves. The results for the pilot regions are presented in Table 5.4.

Table 5.3. Classification of the effective Mesh Size Values for CSI indicator value (Swiss National Park 2014b).

Mesh size	Indicator Value (0–100)
0	0
100	10
250	20
500	30
750	40
1000	50
1250	60
1500	70
2750	80
4000	90
6000	100

Table 5.4. Mean Value for the CSI indicator "fragmentation" in the pilot regions of the Project ECONNECT (Haller et al. 2011). High CSI values stand for bigger mesh size and better connectivity, whereas low CSI values stand for smaller mesh size and higher fragmentation (cf. Table 5.3). The value for the Pilot Region Monte Rosa is not presented in Haller et al. (2011).

Pilot Region	Mean Value for CSI indicator “fragmentation”
Department Isère (including lowland area outside the Alps)	11.0
Southwestern Alps	45
Rhaetian Triangle	56.7
Hohe Tauern Region	62.8
Transboundary Area Berchtesgaden Salzburg	18
Northern Limestone Region	36

Finally, although not an indicator *sensus strictus*, a map of hypothetical barriers and priority conservation areas has been created by ALPARC (Fig. 5.11). The map is not based on a scientific analysis, but was created during a workshop for the Ecological Continuum Initiative and is purely based on expert opinion. Some of the barriers may not be really impermeable (for large carnivores) and in our context less important than presented in Fig. 5.11. The experts did not make allowances for tunnels and bridges, which both reduce the effect of traffic lines as barriers. The experts recommended improving connectivity mainly in the north-south axe of the western French Alps (Durance and Isere Valley), in the west-east orientated Inn valley (Austria), in the north-south orientated axe from Vienna to Graz and Klagenfurt (Austria) and in several sites of the Italian Alps (areas of the big lakes, Susa valley, Assa valley and Val Pusteria; TFPA 2010). Measures mitigating fragmentation could include the protection of non-fragmented areas or special measures for fragmented areas confronted to conflicts of land use. At the time of the analysis, 63.4% of non-fragmented areas occurred within existing protected areas covering about 14,500 km^2 (TFPA 2010).

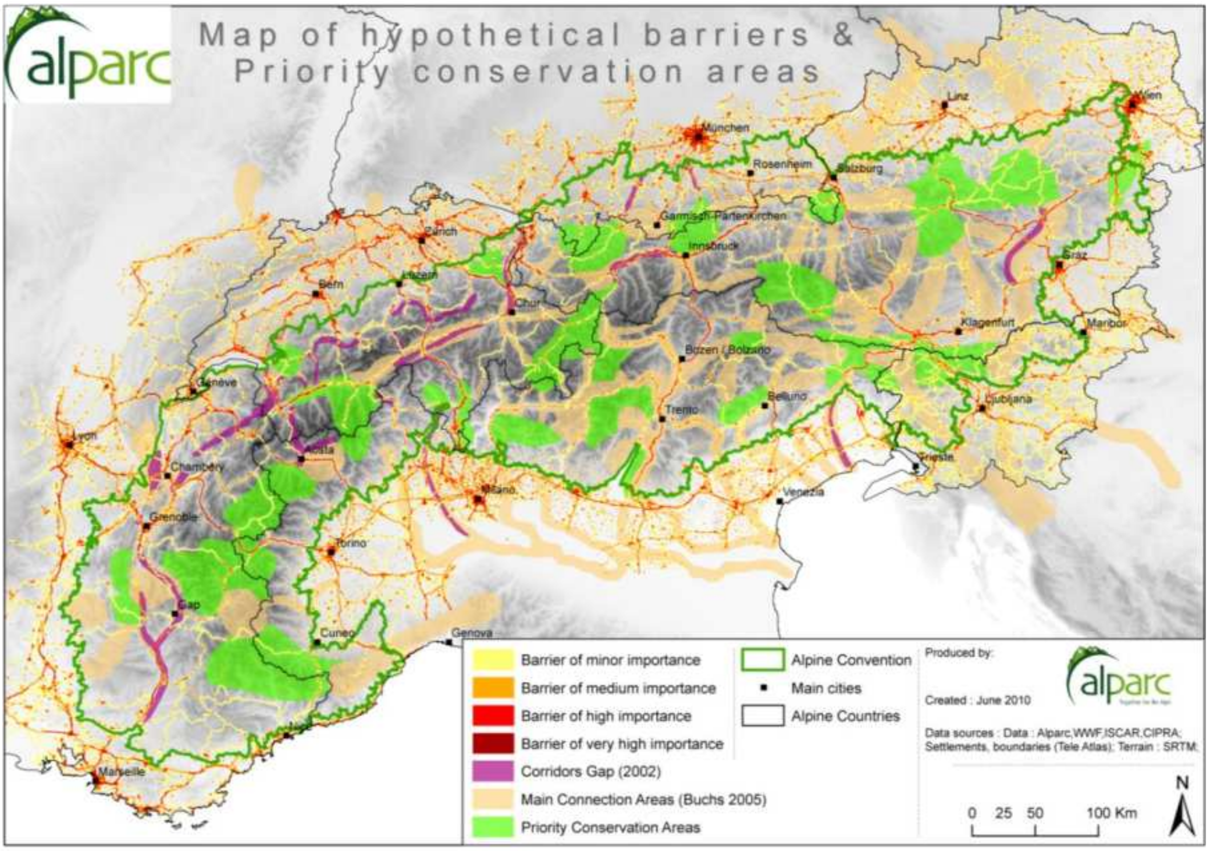


Fig 5.11. Map of barriers and priority conservation areas (Y. Kohler, Alpine Network of Protected Areas ALPARC, pers. comm.). The map was not based on a robust analysis, but was the result of expert opinion expressed during a workshop for the Ecological Continuum Initiative in 2010.

5.3. Availability of wild ungulates (roe deer, red deer, chamois, wild boar)

A key element for the distribution and local abundance of carnivores in the Alps is the availability of prey, hence the prey species spectrum, prey distribution, abundance, and population trends need to be studied and monitored. Wild ungulates are an important source of prey for wolves and lynx in the Alpine region (Chapter 5.5). They are also a key factor determining their return (e.g. success of reintroduction programmes such as the lynx in Switzerland; Breitenmoser 1997; Chapter 3.2).

Populations of all wild ungulate species have been increasing over the past decades and continue to do so in many Alpine regions except for the chamois. This increase was the result of the hunting regimes, but also of the development of human population and the economy (Chapter 5.1). For example, hunting regimes in Italy have influenced the evolution of wild ungulate populations (except those of wild boar) across the country (Apollonio 2004). The rural exodus and the decrease in agricultural exploitation from 67% in 1982 to 42% in 2005 contributed to the increase in wild ungulate distribution across Italy (Apollonio et al. 2010).

The following subchapters summarise available information on the status of wild ungulates which are the key prey species for lynx (roe deer and chamois) and wolf (additionally red deer and wild boar) in the Alpine countries. We do not consider alien species such as mouflon (wild sheep) or fallow deer, as they are only locally distributed in small numbers and are not relevant for the large-scale distribution of wolf or lynx. Some countries make regular records of wild ungulate population sizes available (but often do no state census methods clearly), but others like Austria do not (Reimoser & Reimoser

2010). In these cases we used hunting bag statistics, which are considered to be indicators for long-term changes in populations (Reimoser & Reimoser 2010).

We contacted all of the 24 Italian (autonomous) provinces regarding the availability of hunting bag statistics and/or census numbers for this report. Unfortunately, we only received an answer from the provinces Imperia (G. Torello, pers. comm.), Lecco (R. Facoetti, pers. comm.), Bozen (M. Stadler, pers. comm.), Treviso (S. Busatta, pers. comm.), and Vercelli (S. Raviglione, pers. comm.). The latter referred us to the regional administration from which we did not receive an answer to our subsequent request. As our available data cannot represent the Alpine area of Italy, they are not shown in Figs 5.18, 5.22, 5.29 and 5.36.

5.3.1. Development, distribution and abundance of the red deer in the Alps

France. Red deer populations had strongly declined in the nineteenth century due to loss of habitat and over-hunting, but increased in the 20th century. Hunting bags increased from some 5,000 in 1973 to over 39,000 in 2004, indicating a relatively high abundance also in the departments in the Alps (Fig. 5.12; Maillard et al. 2010).

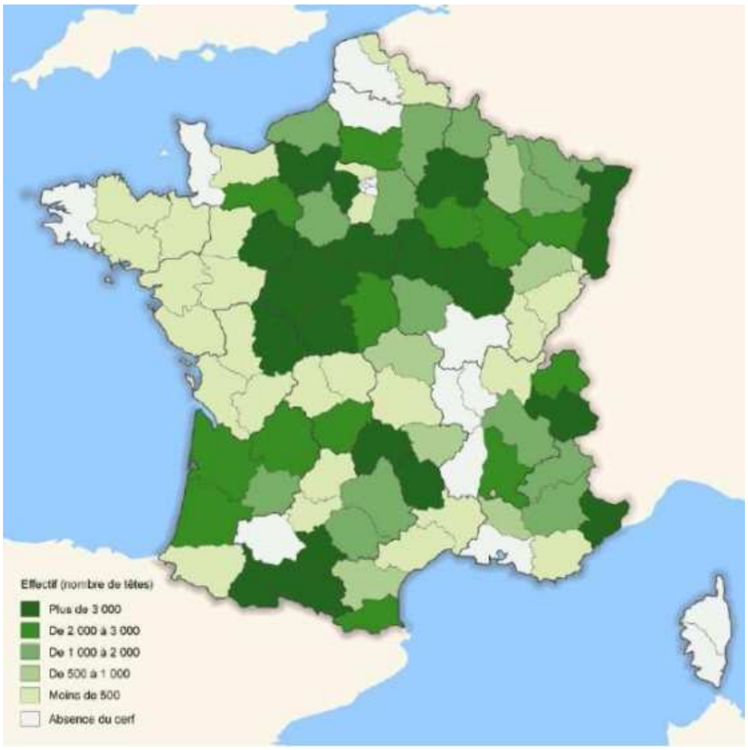


Fig. 5.12. Red deer distribution based on hunting bag data showing population sizes per department across France (ONCFS 2009).

Italy. Most red deer populations in Italy were reintroduced between 1950 and 2003 (Apollonio et al. 2010) and subsequently colonised the Italian Alps and spread into neighbouring countries. Carnevali et al. (2009) mapped out the distribution of red deer in Italy and also indicated densities for the various provinces with census estimates for the Alpine regions at 49,074 in 2005 (Fig 5.13).

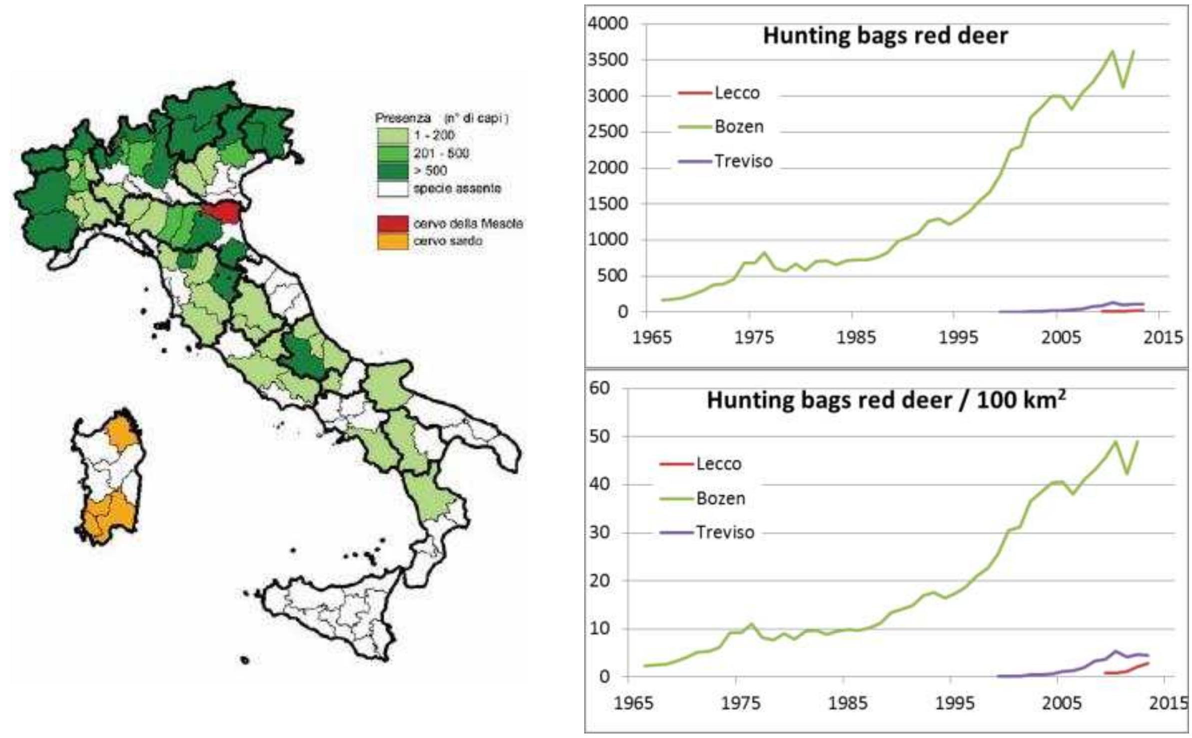


Fig. 5.13. Left: Distribution of red deer (number of heads) in the Italian provinces in 2005. Orange= provinces with Sardinian Red Deer, red= province with autochthonous Mesola Wood population, white = provinces where the species is absent (Carnevali et al. 2009). Top right: Hunting bags red deer in three Italian provinces. Bottom right: Hunting bags red deer per 100 km^2 in three Italian provinces (R. Facoetti, pers. comm., M. Stadler, pers. comm., S. Busatta, pers. comm.).

Switzerland. Red deer went extinct in Switzerland in the 1850s (Breitenmoser & Breitenmoser-Würsten 2008). Natural recolonisation from the Montafon region in Austria, supported by releases in different areas let the red deer population recover (Fig. 5.14). The highest densities are found in the eastern (e.g. Canton of Grisons) and southern (e.g. Canton of Valais) parts of the Swiss Alps, whereas the population continues to increase in the north-western Alps. As of 2013, the Swiss red deer population was estimated at 33,552 (BAFU 2014) with the majority of them present in the Alps.

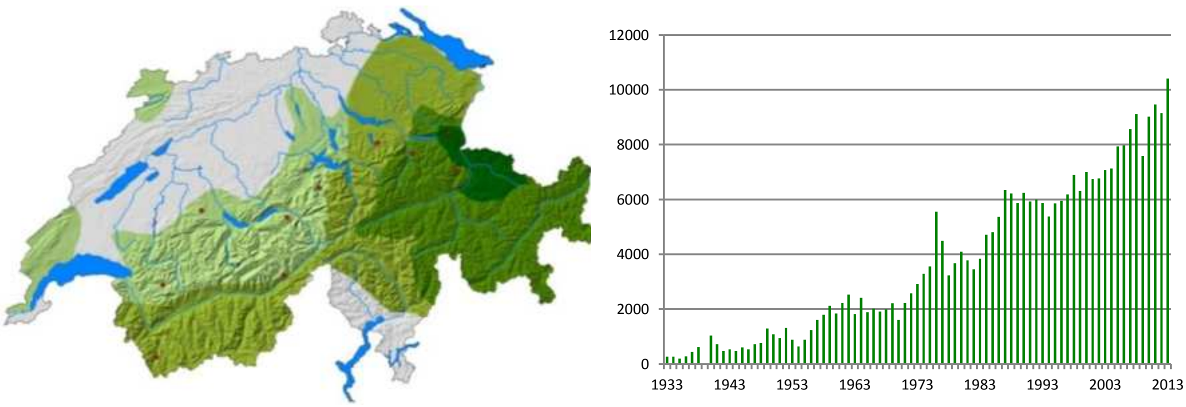


Fig. 5.14. Left: The return of the red deer to Switzerland from Austria. From dark to light green: 1900, 1936, 1961, 1982. Red dots: known reintroduction sites. Map from Breitenmoser & Breitenmoser-Würsten 2008. Right: Evolution of the hunting bag of red deer in Switzerland (BAFU 2014).

Liechtenstein. Red deer hunting bags in Liechtenstein increased from 185 to 218 between 1993 and 2013 (Wolfgang Kersting, Amt für Umwelt, Liechtenstein, pers. comm.). Red deer was mainly hunted on the slopes at medium altitude, where most of the protective forests are located (Fig. 5.15).

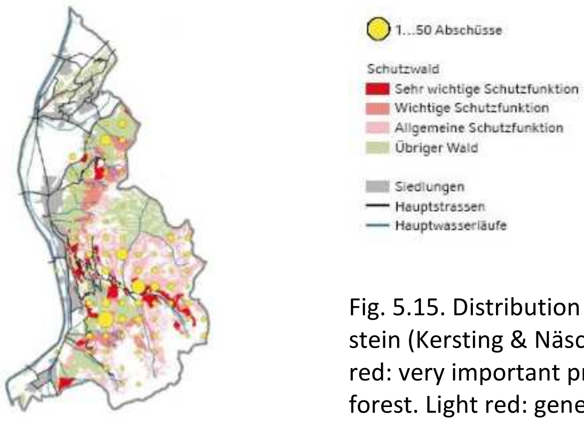


Fig. 5.15. Distribution of the red deer hunting bag 2004–2007 in Liechtenstein (Kersting & Näscher 2008). Yellow circles indicate shooting sites. Dark red: very important protective forest. Medium red: important protective forest. Light red: general protective forest. Green: other forest.

Germany. In the past, red deer was an important game species and special efforts were taken to raise the population in the royal hunting grounds of Bavaria (Wotschikowsky 1998). These populations experienced a severe decline during the 19th century due to over-hunting (Kuehn et al. 2003). Populations then recovered in specially designated protected areas. Since the 1960s the annual harvests increased from 25,000 to over 50,000 individuals; however, the distribution is not even across the country. Red deer populations in Germany occupy about 23% of the country, mostly the large state forests, which are specific red deer areas. Deer beyond these zones have to be eliminated (apart from a few exceptions) to protect the forest from browsing damage (Wotschikowsky 2010). The German Alps are a typical and high-density red deer zone (Fig. 5.16, Deutsche Wildtier Stiftung 2014).

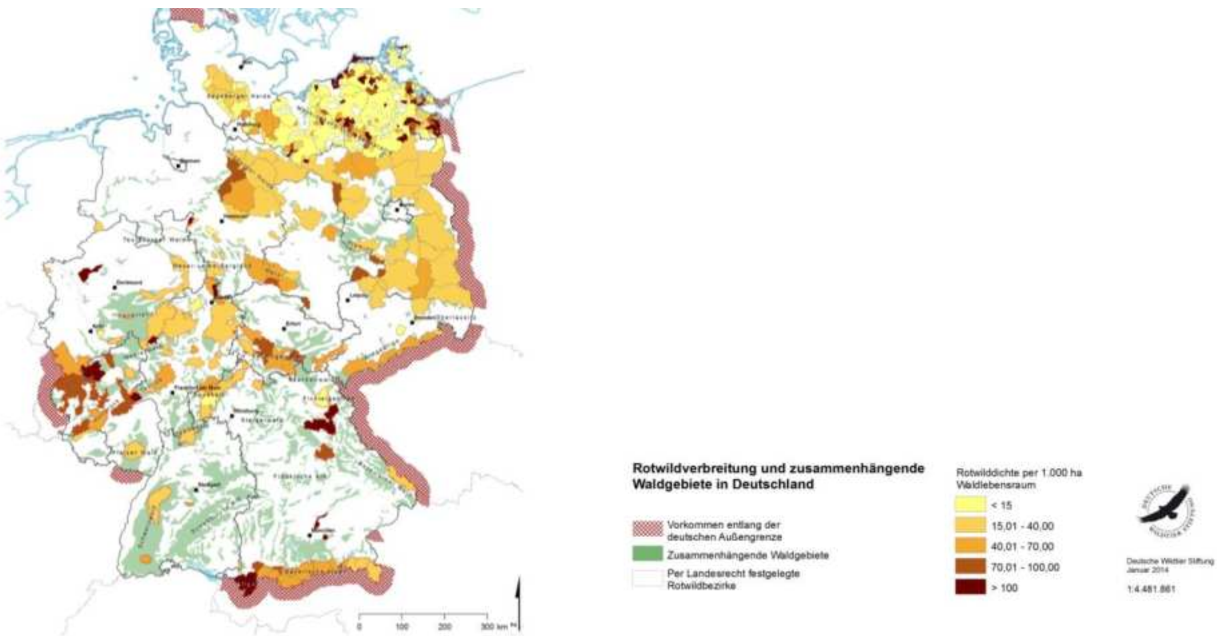


Fig. 5.16. Red deer density (red deer/10 km² forest habitat) in Germany (Deutsche Wildtierstiftung 2014).

Austria. Hunting in the past had a negative impact on the populations of red deer in Austria. Following changes in management practises, the red deer population increased and with it the hunting bag. The increasing numbers also allowed the population to expand into neighbouring countries, as for example also into eastern Switzerland. Today red deer are hunted across the Austrian Alps (Fig. 5.17, Fig. 5.18a).

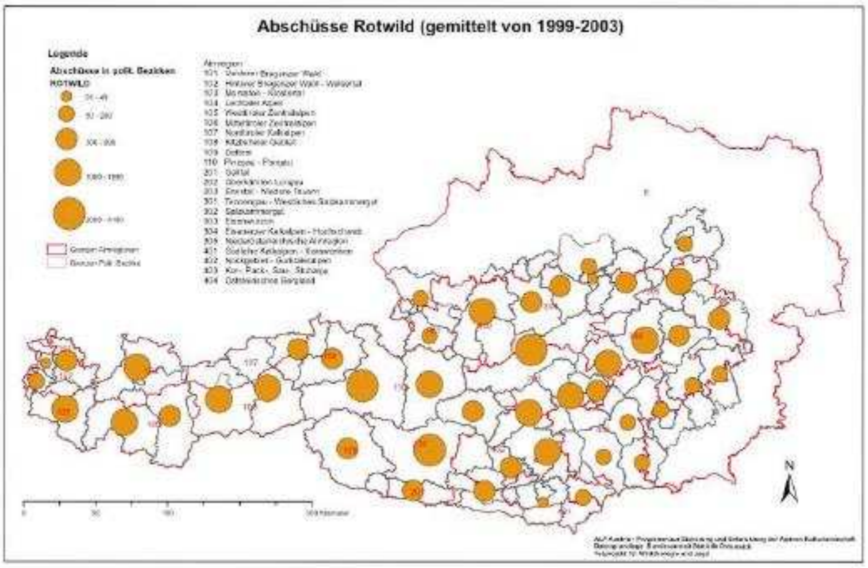
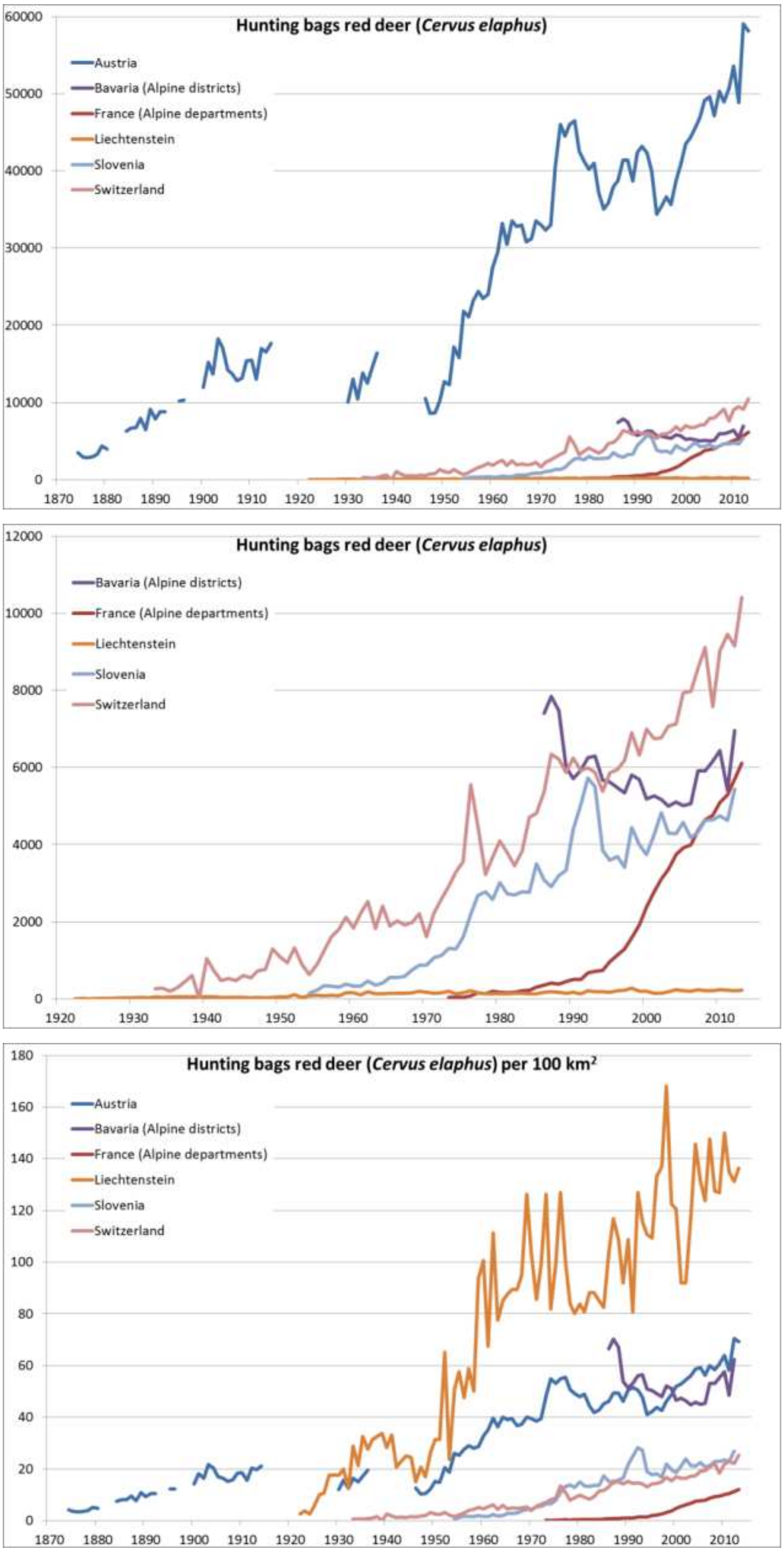


Fig. 5.17. Hunting bag of red deer in the Austrian Alps, averaged over the years 1999-2003. Increasing orange dots represent 1-49, 50-299, 300-999, 1000-1999, 2000-4100 animals shot. Red polygons = Alm regions, gray polygons = political boundaries (Huber & Bergier 2006).

Slovenia. Red deer populations in Slovenia fluctuated greatly in the last two centuries (Jerina & Adamic no date). From 1907 to 1938, red deer population was collected based on harvest, road kills and dead animals that were found opportunistically (Klopcic et al. 2010). After 1948, these data were collected in a systematic manner and maintained by the official hunting register. After 1976, the quality of data collected was improved with more detailed information such as age, sex, etc., being recorded. The highest red deer density was recorded in 1977 (5.8 km^{-2}) and the lowest in 1987 (2.5 km^{-2}; Klopcic et al. 2010). Red deer can be found in about 80% of Slovenian forests as a result of combined efforts to improve habitat and feeding as well as a strengthening legal protection. The population size in 2010 was estimated to be between 10,000 and 14,000 individuals (Adamic & Jerina 2010). The red deer population shows an increasing trend in Slovenia in the last decades, both in abundance and spatial distribution. In 2014, hunting bags reached a new record for red deer with 6,344 animals (M. Jonozovič, pers. comm.). The population is also significantly increasing in the Alpine region. Hunting bags in the three Alpine Hunting Management Districts averaged over the last ten years 1,087 individuals (Min: 904 in 2007; Max: 1,239 in 2013; M. Jonozovič, pers. comm.).

Hunting bags for red deer across all the Alpine countries show an increasing trend with strong increases in Austria as compared to the other countries. The following graphs show hunting bag data for the Alpine countries (Fig. 5.18a), hunting bag data without Austria (Fig. 5.18b) and data per 100 km² (Fig. 5.18c). Data for Austria was removed in Fig. 5.18b to provide an overview of the other countries using an adapted scale.



a) Hunting bags for red deer. Data for whole countries if not indicated otherwise.

b) Hunting bags for red deer (without Austria). Data for whole countries if not indicated otherwise.

c) Hunting bags for red deer per 100 km^2. Data for whole countries if not indicated otherwise.

Fig. 5.18. Hunting bag statistics for red deer. Sources: AT: until 1982: see Breitenmoser & Breitenmoser-Würsten 2008; 1983-2013: Statistik Austria 2014; FR: Data ´Réseau Ongulés Sauvages´, ONCFS/FNC/FDC; DE: Reinhard Menzel, Bayerisches Staatsministerium für Ernährung, Landwirtschaft und Forsten, pers. comm., Friedrich Pielok, Bayerisches Staatsministerium für Ernährung, Landwirtschaft und Forsten, pers. comm., Frank Tottewitz, Thünen-Institut für Waldökosysteme, pers. comm.; FL: Wolfgang Kersting, Amt für Umwelt, Liechtenstein, pers. comm.; SL: Statistical Office of the Republic of Slovenia 2014a; CH: BAFU 2014.

5.3.2. Development, distribution and abundance of the roe deer in the Alps

France. Roe deer occur all over the continental country (Fig. 5.19a, Réseau Ongulés Sauvages, ONCFS/FNC/FDC), occupying agricultural landscapes as well as mountainous habitats (Maillard et al. 2010). Before 1979, the different hunting societies collected hunting statistics for their areas making it difficult to calculate accurate estimates (Boisaubert et al. 1999). In 1979, an annual standardised hunting plan was made and data were collected in a systematic way across the country. The roe deer population increased between 1980 and 1998. Using population census data, Boisaubert et al. (1999) estimated the roe deer population in 1997-1998 at over a million individuals. In the French Alps the hunting bag of roe deer increased until the early 2000s, after that it levelled off (Fig. 5.19b).

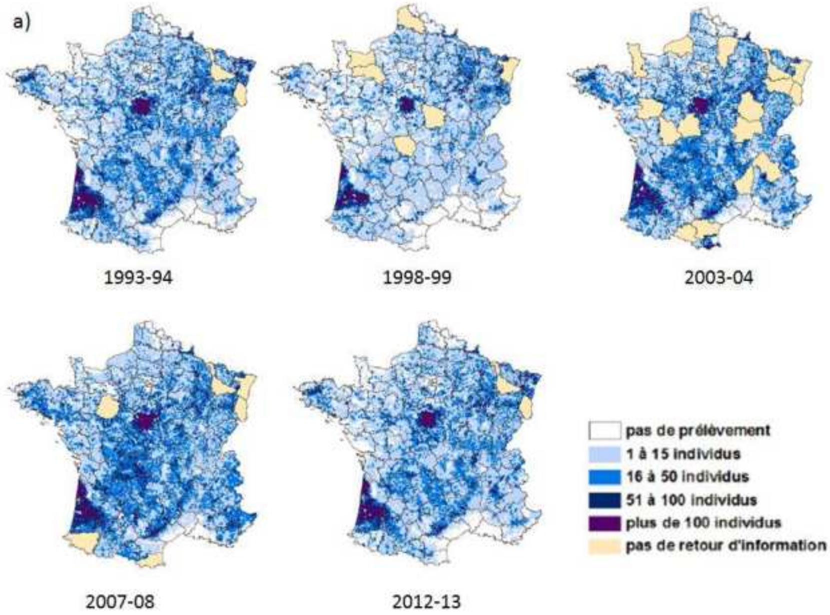


Fig. 5.19a) Development and distribution of roe deer hunting bags per municipality (Data ´Réseau Ongulés Sauvages´, ONCFS/FNC/FDC). The colours from white to dark blue indicate the number of animals removed for the respective hunting season: 0, 1-15, 16-50, 51-100, >100. Pale yellow = no data.

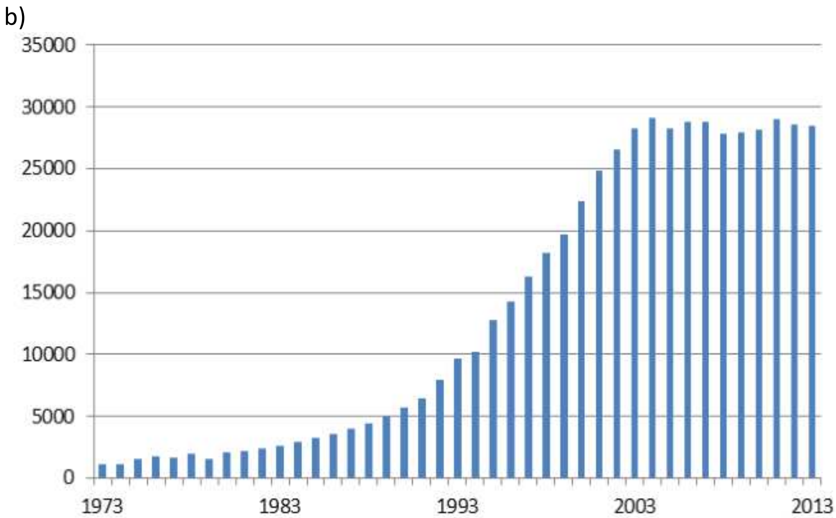


Fig. 5.19b) Evolution of the number of killed roe deer in the French Alps (Data ´Réseau Ongulés Sauvages´, ONCFS/FNC/FDC).

Italy. Between 1980 and 1998, there was an increase of about 60% of the roe deer population (Apollonio 2004). The roe deer population in Italy is composed of a combination of native animals, reintroduced animals and individuals that immigrated from the northern Alps (Apollonio et al. 2010). This has given rise to concerns about a possible genetic compromise in the population of the native subspecies (*Capreolus capreolus italicus*). Roe deer populations are distributed in different regions across the country and are present in different densities in the various provinces (Fig 5.20; Mattioli et al. 2004). In 2005, the roe deer population in the Italian Alps was estimated at 184,260 individuals (Carnevali et al. 2009).

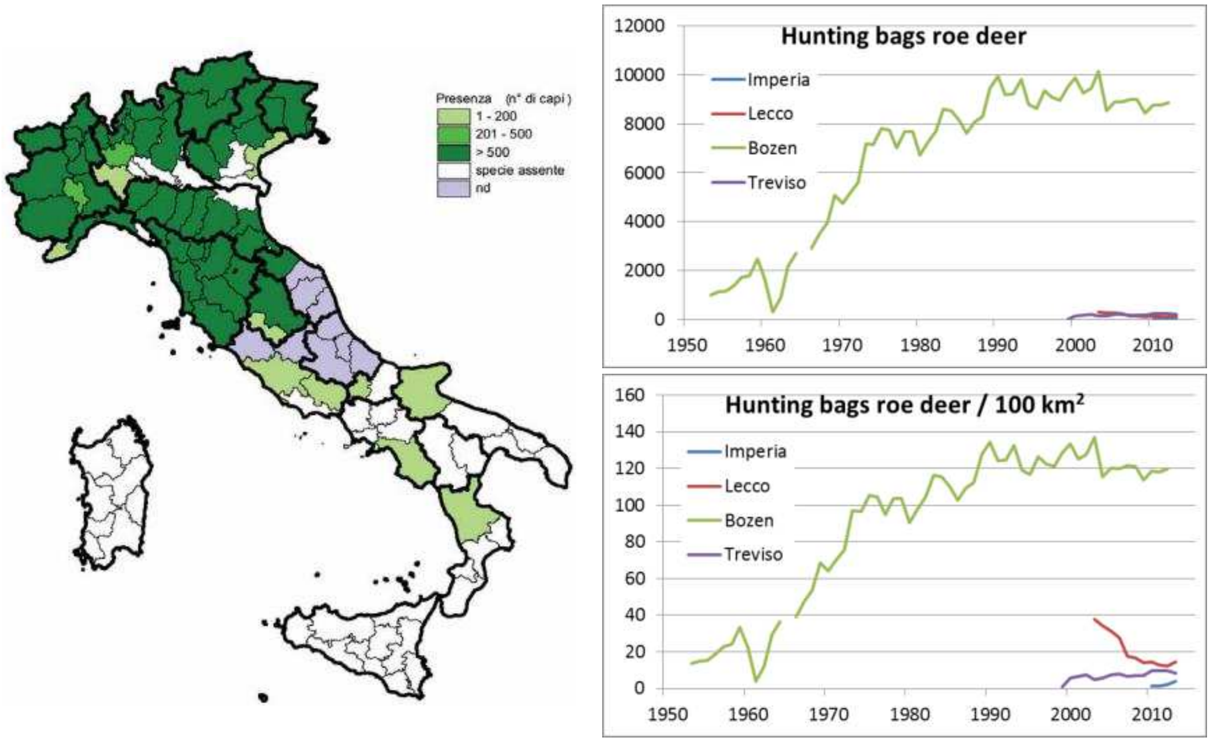


Fig 5.20. Left: Roe deer distribution (number of heads) in the Italian provinces in 2005. Purple = deer presence, but no abundance data available, white= areas where the species is absent (Carnevali et al. 2009). Top right: Hunting bags roe deer in four Italian provinces. Bottom right: Hunting bags roe deer per 100 km^2 for four Italian provinces (G. Torello, pers. comm., R. Facoetti, pers. comm., M. Stadler, pers. comm., S. Busatta, pers. comm.).

Switzerland. The roe deer has been rare in Switzerland already during the 17th and 18th century. Probably it was never completely gone. There were several attempts to reintroduce the species, as early as the second half of the 18th century. It is not clear how much these efforts have contributed to the recovery of the roe deer. More relevant probably was the expansion of the roe deer population of Baden-Württemberg at the end of the 19th century (Fig. 5.21). Today the roe deer is the most abundant wild ungulate in the country occurring up to the timberline with a stable population which is estimated at 138,452 individuals in 2013 (BAFU 2014).

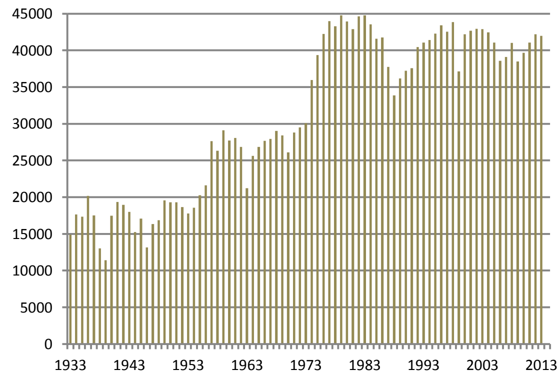


Fig. 5.21. Left: The return of the roe deer to Switzerland from southern Germany. From dark to light brown: situation around 1900, 1920, 1930. Map from Breitenmoser & Breitenmoser-Würsten 2008. Right: Evolution of the hunting bag of roe deer in Switzerland (BAFU 2014).

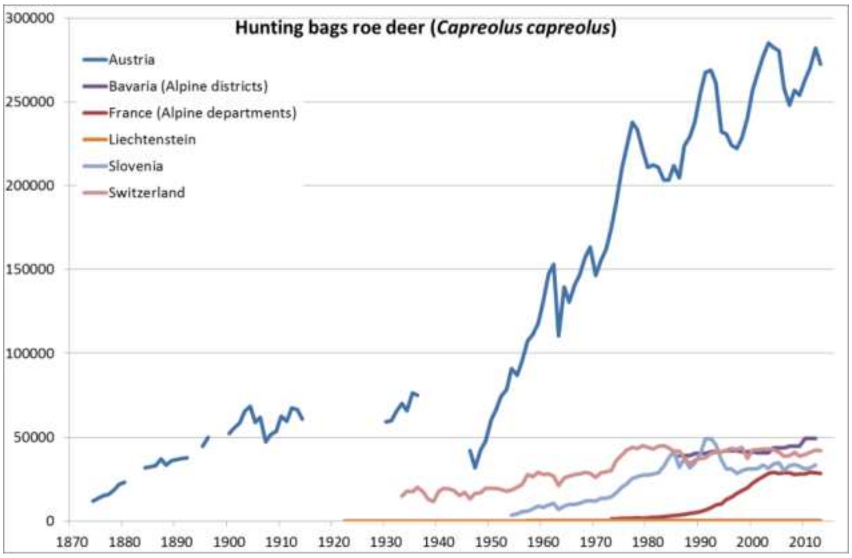
Liechtenstein. In Liechtenstein, roe deer populations are low and appear relatively stable with hunting bag numbers of 217 in 1993 and 202 in 2013 (Wolfgang Kersting, Amt für Umwelt, pers. comm.).

Germany. Roe deer are present across all of Germany, up to an elevation of 1800 m (Wotschikowsky 2010). They occur in a variety of habitats ranging from natural habitats to human dominated landscapes.

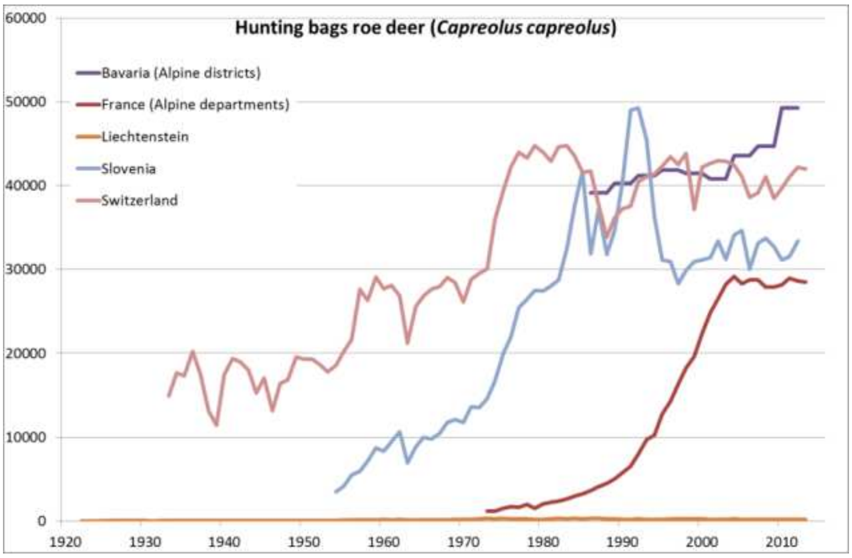
Austria. In Austria, roe deer hunting bag has increased steadily over the years with a net increase in mortality caused by road kills (Reimoser & Reimoser 2010).

Slovenia. Roe deer are the most abundant wild ungulate species in Slovenia, with a stable population and hunting bags of over 41,000 individuals in some years (M. Jonozovič, pers. comm.). Hunting bags in the three Alpine Hunting Management Districts averaged over the last ten years 7,804 individuals (Min: 7,483 in 2011; Max: 8,454 in 2005; M. Jonozovič, pers. comm.).

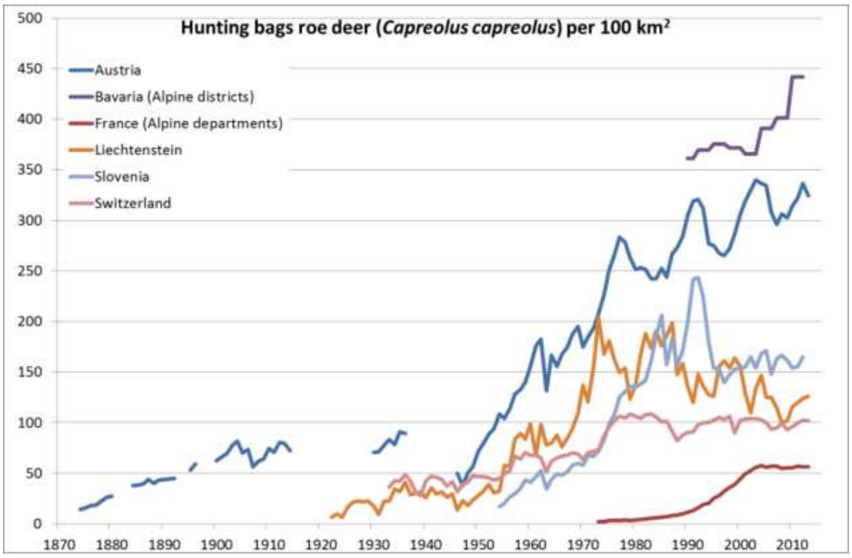
Roe deer hunting bags are stable or show an increasing trend in most Alpine countries with a distinct increase in Austria and Bavaria. The following graphs show the evolution of roe deer hunting bags across the Alpine countries. The first figure shows all countries (Fig. 5.22a), the second shows data for all countries except Austria (Fig. 5.22b) and the third shows data per 100 km² (Fig. 5.22c).



a) Hunting bags for roe deer. Data for whole countries if not indicated otherwise.



b) Hunting bags for roe deer (without Austria). Data for whole countries if not indicated otherwise.

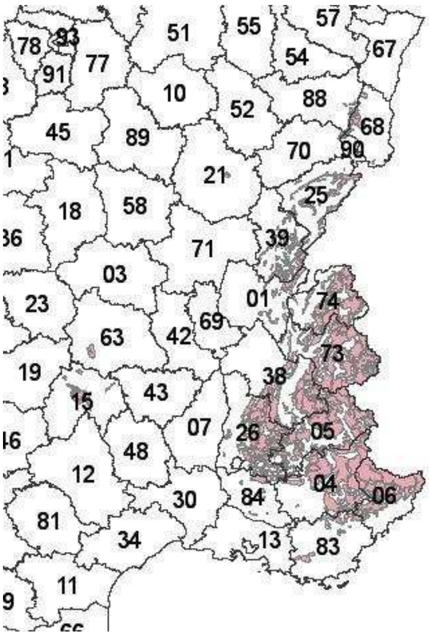


c) Hunting bags for roe deer per 100 km^2. Data for whole countries if not indicated otherwise.

Fig. 5.22. Hunting bag statistics for roe deer. Sources: AT: until 1982: see Breitenmoser & Breitenmoser-Würsten 2008; 1983-2013: Statistik Austria 2014; FR: Data ´Réseau Ongulés Sauvages´, ONCFS/FNC/FDC; DE: Reinhard Menzel, Bayerisches Staatsministerium für Ernährung, Landwirtschaft und Forsten, pers. comm., Friedrich Pielok, Bayerisches Staatsministerium für Ernährung, Landwirtschaft und Forsten, pers. comm., Frank Tottewitz, Thünen-Institut für Waldökosysteme, pers. comm.; FL: Wolfgang Kersting, Amt für Umwelt, Liechtenstein, pers. comm.; SL: Statistical Office of the Republic of Slovenia 2014a; CH: BAFU 2014.

5.3.3. Development, distribution and abundance of the chamois in the Alps

France. The distribution and population evolution of chamois in France has not been well documented. In the beginning of the 19th century, when the human population was at its highest in the Alpine region and due to agro-forestry practises (Chapter 5.1, 5.2), the chamois were thought to have found refuge at higher altitudes (Muséum national d'Histoire naturelle 2014). Their populations were thought to have increased substantially before falling after the First World War. Following the Second World War, the chamois populations experienced a decline as a result of heavy hunting pressure (Fayard 1984). With the establishment of national parks and protected areas, the chamois population subsequently recovered. About 60,000 chamois are currently present in the French Alps (Maillard et al. 2010) following a well-managed population intervention through a controlled harvest (Fig. 5.23).



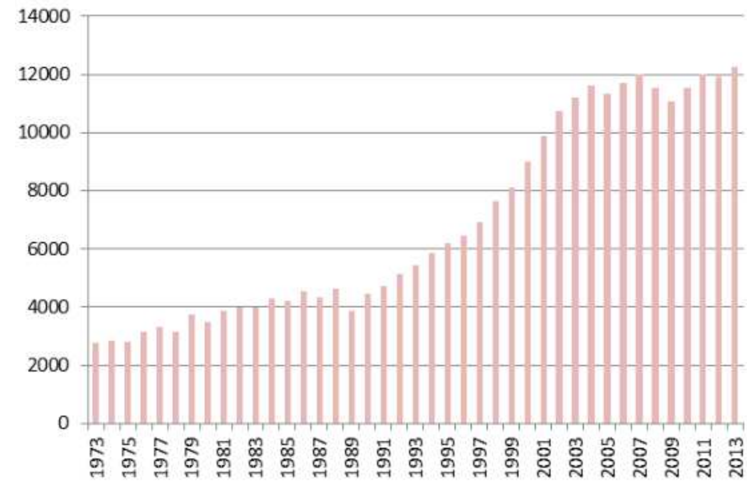


Fig. 5.23. Left: Distribution of the chamois in France 2010. Right: Evolution of the hunting bag of chamois in France 1973-2013 (Data ´Réseau Ongulés Sauvages´, ONCFS/FNC/FDC).

Italy. The chamois is thought to occur across the Italian Alps and their populations increased slightly following minor translocation efforts to the western Alps (Fig. 5.24; Apollonio et al. 2010). These translocations were carried out using individuals from the Gran Paradiso National Park in Italy; however, these individuals had different mitochondrial lineages (Apollonio et al. 2010). The exact effect and impact of such reintroductions need to be studied in depth to determine if there are any consequences (Apollonio 2004). Human expansion and pressure was reported to have little impact on this species (Apollonio 2004). Census data show that there were about 131,714 individuals in 2010 (Raganella Pelliccioni et al. 2013), with about one-third of the population being found in the Trentino-Alto Adige region (Carnevali et al. 2009). Dupré et al. (2001) have mapped out the chamois distribution across the Italian Alps indicating densities (Fig. 5.24).

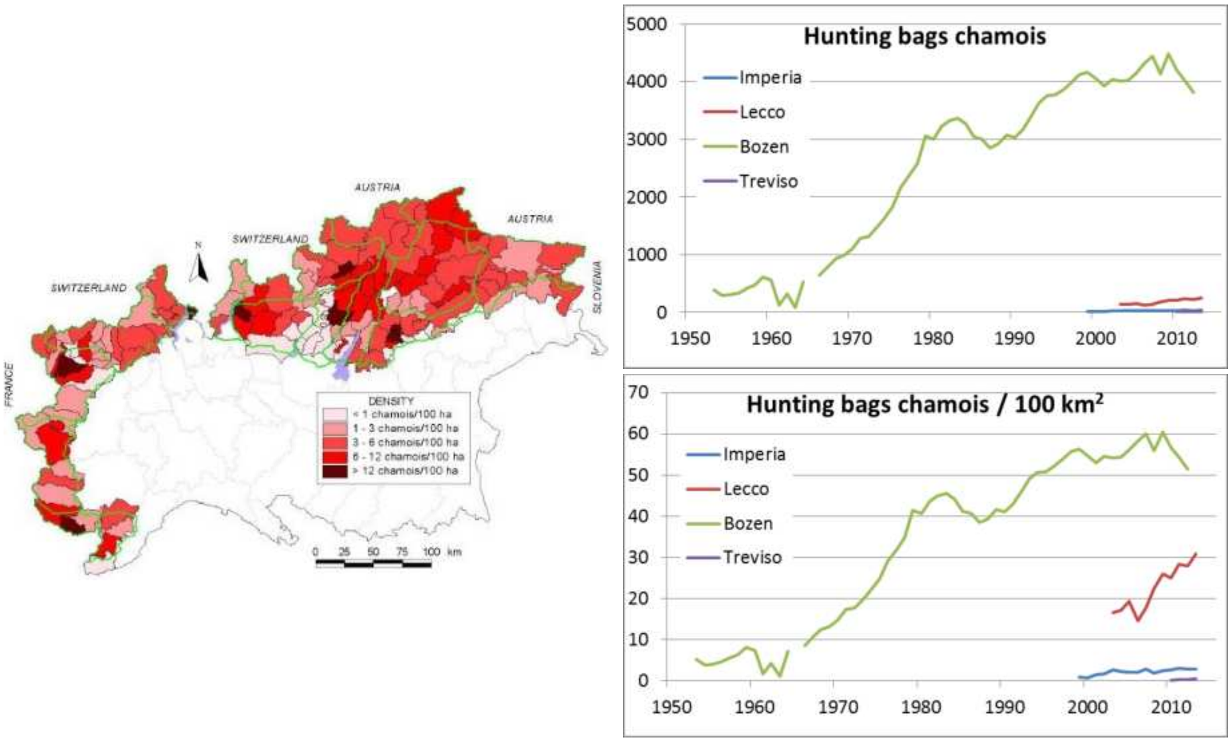


Fig. 5.24. Left: Distribution and density of chamois populations in the Italian Alps with the different colours indicating the intensity of these densities (Dupré et al. 2001). Top right: Hunting bags chamois in four Italian provinces. Bottom right: Hunting bags per 100 km^2 for four Italian provinces (G. Torello, pers. comm., R. Facoetti, pers. comm., M. Stadler, pers. comm., S. Busatta, pers. comm.).

Switzerland. Chamois populations in Switzerland were never fully exterminated even at the turn of the century. However, they were relatively low in numbers. Following the enactment of the first federal hunting law in 1875, the chamois population recovered and established itself across suitable habitats across the Alps and Prealps (Fig 5.25; Imesch-Bebié et al. 2010). It increased until around 1995 after which the numbers started to decrease. In 2013, the chamois population was estimated to be 90,803 individuals and to be still decreasing (BAFU 2014).

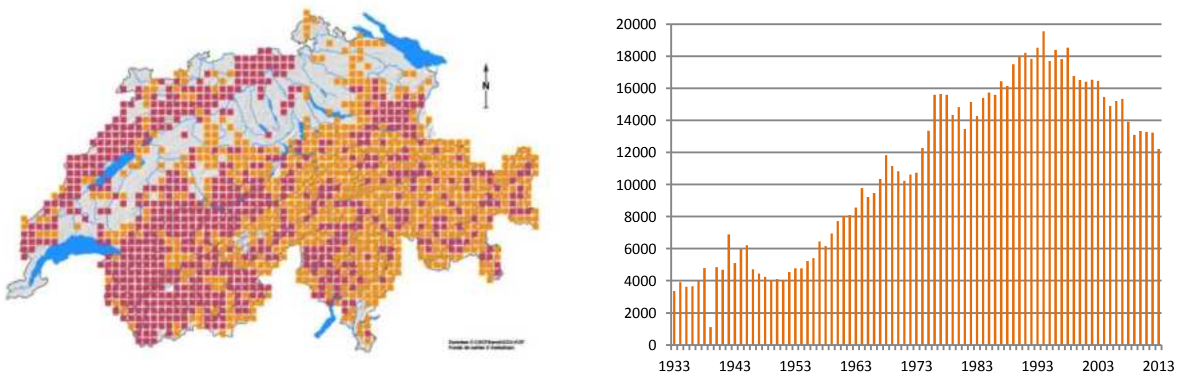


Fig. 5.25. Left: Distribution of chamois in Switzerland. Orange squares: confirmed presence up to the year 2000, purple squares: confirmed presence since 2000. (CSCF 2014). Right: Evolution of the hunting bag of chamois in Switzerland (BAFU 2014).

Liechtenstein. Data for chamois hunting bags in Liechtenstein indicate harvests of 109 in 1993 and 152 in 2013 (Wolfgang Kersting, Amt für Umwelt, pers. comm.).

Germany. Chamois are mainly present in the Bavarian Alps of Germany and the Black Forest (Fig. 5.26; Wotschikowsky 2010). This chamois population is connected with larger populations in Austria, Switzerland, France and Northern Italy. Annual harvests in the 1960s were at 2000 individuals. Harvests increased to about 4000 individuals annually in 2010 (Wotschikowsky 2010).

Fig. 5.26. Distribution of chamois in Germany (Briedermann et al. 1997).

Austria. Chamois populations in Austria were subjected to high hunting pressure, especially during the 1980s and 1990s (Reimoser & Reimoser 2010). As estimates of wild ungulate populations are not available for this country, it is difficult to determine population trends for species such as the chamois. Estimated growth rate based on registered mortality indicate a slight decrease in the chamois population (Reimoser & Reimoser 2010). Hunting bag data for the period between 1999 and 2003 for the whole country was represented graphically (Fig. 5.27).

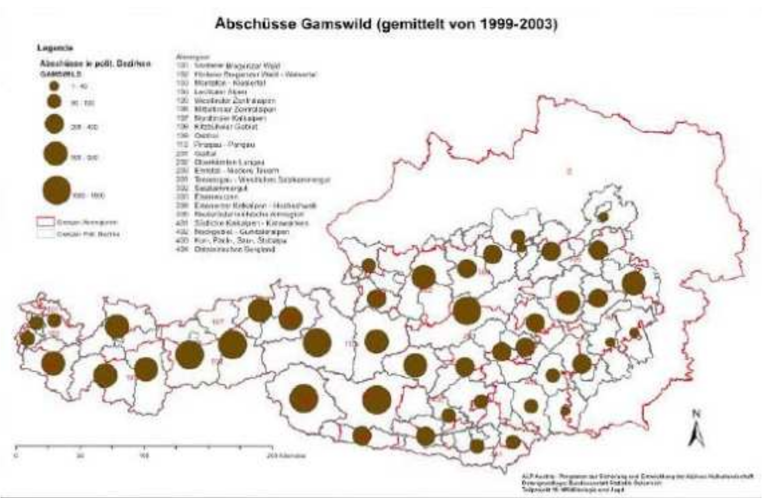


Fig. 5.27. Hunting bag of chamois in the Austrian Alps, averaged over the years 1999-2003. Increasing brown dots represent 1-49, 50-199, 200-499, 500-999, 1000-1500 animals shot. Red polygons = Alm regions, gray polygons = political boundaries (Huber & Bergier 2006).

Slovenia. Chamois are native to the Slovenian Alps and Dinaric mountains (Adamic & Jerina 2010). They were widespread across the mountainous habitats of Slovenia at the beginning of the twentieth century (Adamic & Jerina 2010). Although they were widespread, the populations were small and isolated. Restocking through translocations in 1927, 1957 and 1959 and spontaneous recolonisation alongside legal protection along with a lack of natural predators contributed to the strengthening of the chamois population (Adamic & Jerina 2010). However, in the years following 1975 an outbreak of sarcoptic mange affected a large portion of the population (Chapter 6.2.2). About 80% of the stock was lost (Fig. 5.28; Adamic & Jerina 2010). In 2005, the hunting bag data included 2,506 chamois (Adamic & Jerina 2010) and the population shows a slightly decreasing trend (M. Jonozovič, pers. comm.). Hunting bags in the three Alpine Hunting Management Districts averaged over the last ten years 1,622 individuals (Min: 1,465 in 2014; Max: 1,801 in 2006; M. Jonozovič, pers. comm.).

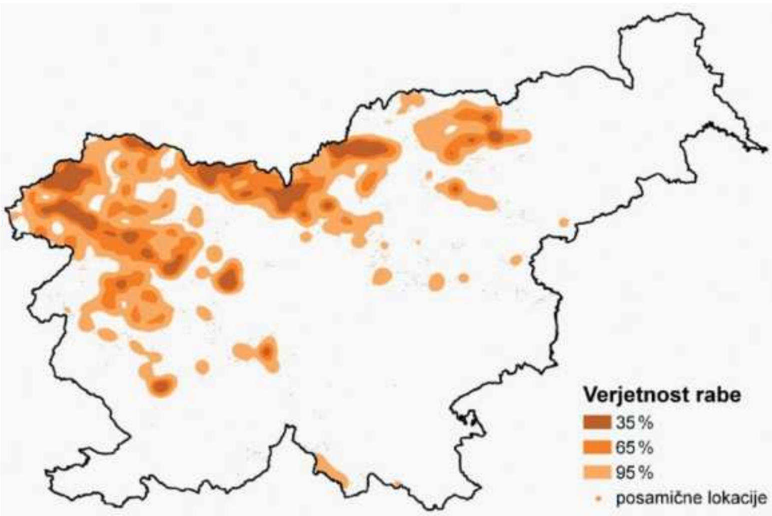
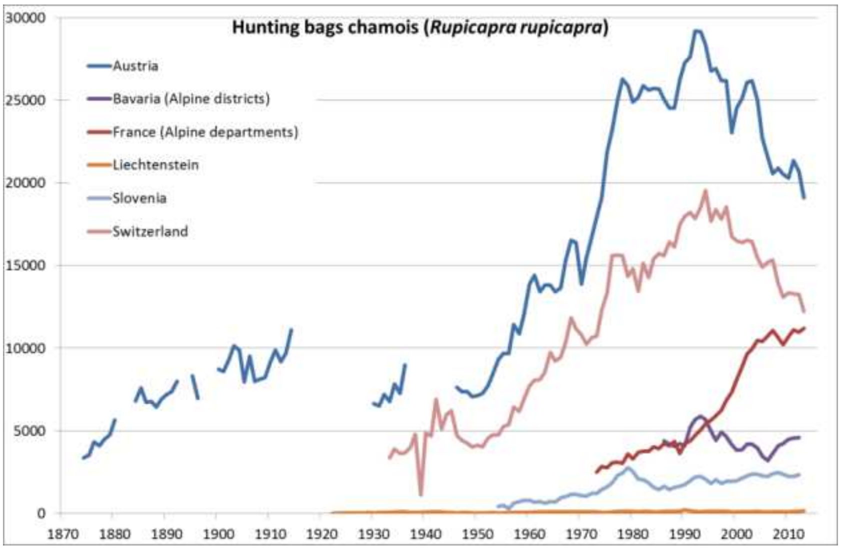
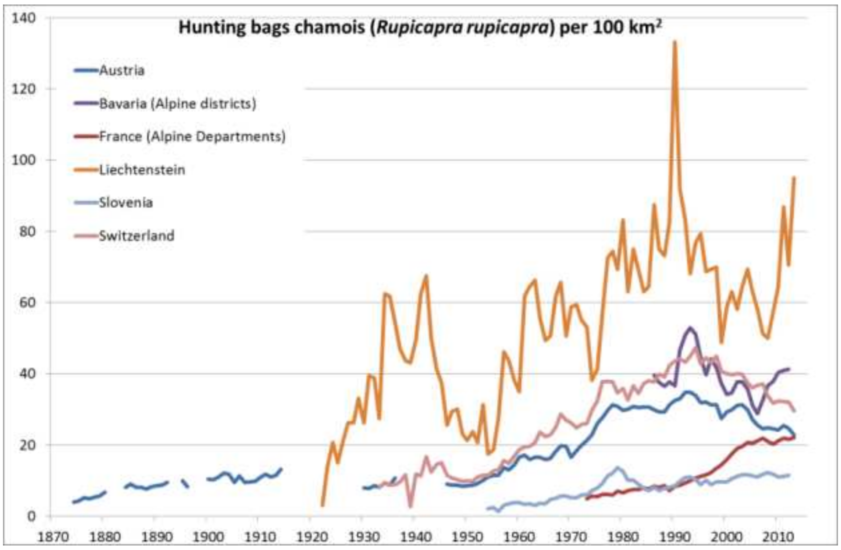


Fig. 5.28. Distribution range of chamois in Slovenia, estimated based on data from the 'Central Slovene Register of Large Game Species and Large Carnivores' (Stergar et al. 2009). Coloured areas show (cumulatively) 35, 65 and 95%, respectively, polygons enclosing the locations of all harvested chamois in Slovenia, while points delineate individual locations of harvested chamois (Apollonio et al. 2010).

Hunting bags indicate an increase in harvests until the 1990s followed by a decrease in the hunting bag in most of the countries except France (Fig. 5.29). The following graphs show the hunting bag data for all the Alpine countries (Fig. 5.29a), and data per km² (Fig. 5.29b).



a) Hunting bags for chamois. Data for whole countries if not indicated otherwise.



b) Hunting bags for chamois per 100 km^2. Data for whole countries if not indicated otherwise.

Fig. 5.29. Hunting bag statistics for chamois. Sources: AT: until 1982: see Breitenmoser & Breitenmoser-Würsten 2008; 1983-2013: Statistik Austria 2014; FR: Data ´Réseau Ongulés Sauvages´, ONCFS/FNC/FDC; DE: Reinhard Menzel, Bayerisches Staatsministerium für Ernährung, Landwirtschaft und Forsten, pers. comm., Friedrich Pielok, Bayerisches Staatsministerium für Ernährung, Landwirtschaft und Forsten, pers. comm., Frank Tottewitz, Thünen-Institut für Waldökosysteme, pers. comm.; FL: Wolfgang Kersting, Amt für Umwelt, Liechtenstein, pers. comm.; SL: Statistical Office of the Republic of Slovenia 2014a; CH: BAFU 2014.

5.3.4. Development, distribution and abundance of the wild boar in the Alps

France. Wild boar have become widespread across France (Fig. 5.30), with the numbers of individuals hunted having increased twelve-fold from 36,429 in 1973 to 443,578 in 2004 (Maillard et al. 2010). Their numbers rose slowly but steadily until 1989 and then increased significantly (Maillard et al. 2010). In the French Alps, the hunting bag started to increase in the early 1990s and peaked in 1999-2001. Afterwards it decreased considerably, maybe reflecting the arrival and spreading of the wolf in the Alps (Fig. 5.31).

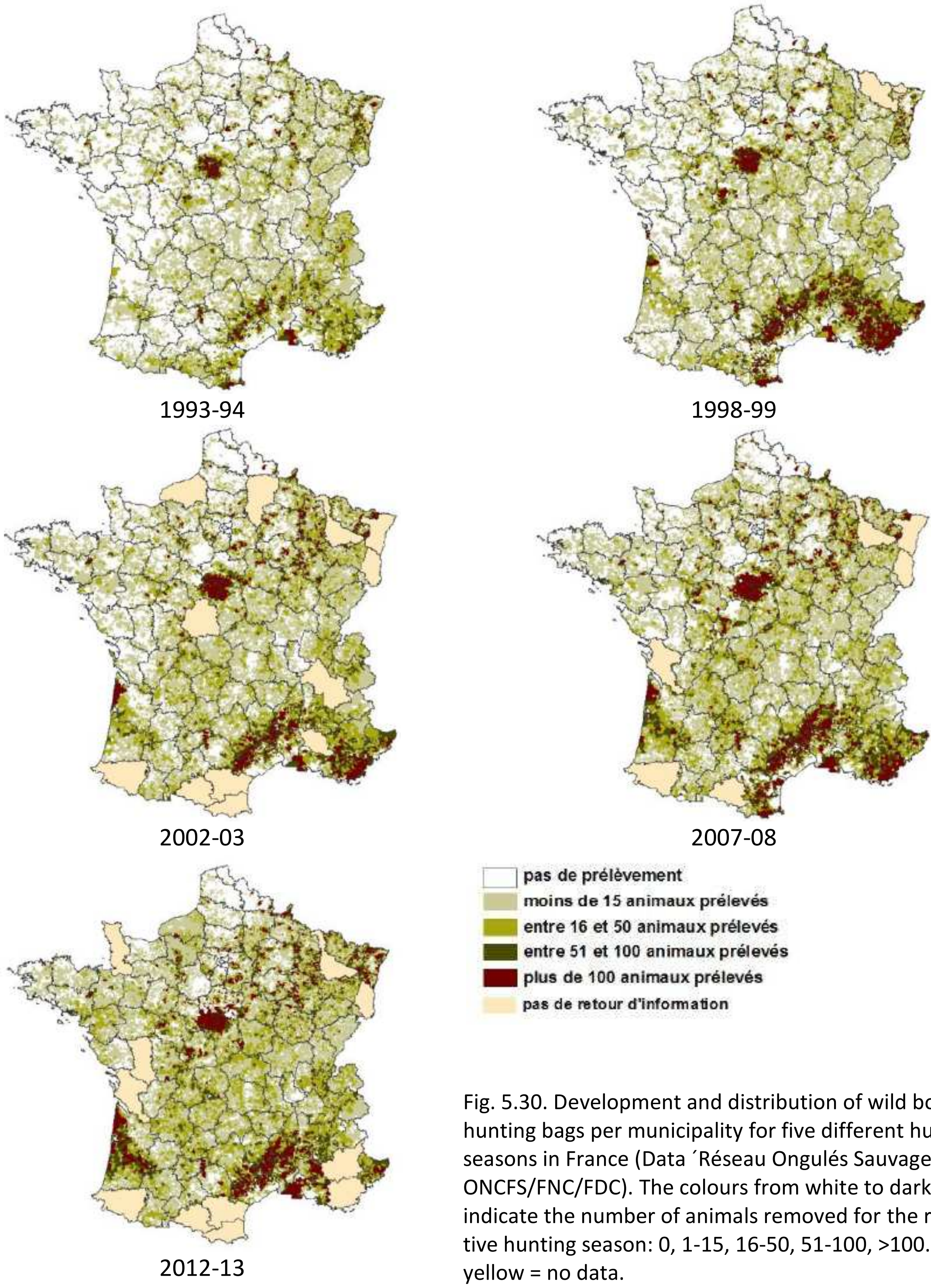


Fig. 5.30. Development and distribution of wild boar hunting bags per municipality for five different hunting seasons in France (Data ´Réseau Ongulés Sauvages´, ONCFS/FNC/FDC). The colours from white to dark brown indicate the number of animals removed for the respective hunting season: 0, 1-15, 16-50, 51-100, >100. Pale yellow = no data.

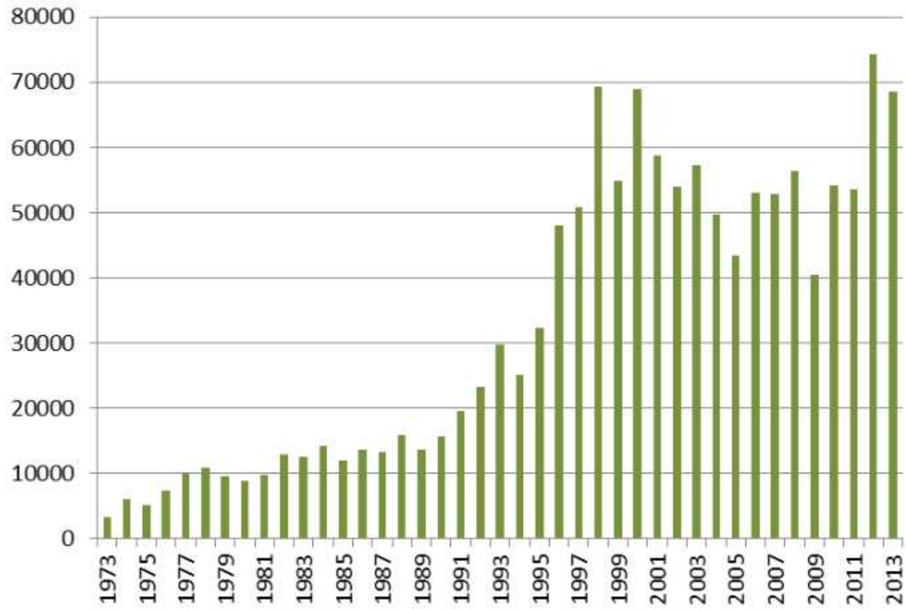


Fig. 5.31. Evolution of the hunting bag of wild boar in the French Alps (Data ´Réseau Ongulés Sauvages´, ONCFS/FNC/FDC).

Italy. Wild boar populations in Italy are thought to be genetically compromised due to the mismanagement of the native population (Apollonio et al. 2010). Currently, they are widespread across the country (Apollonio et al. 2010). Although there is no census data for wild boar estimates for the country, Carnevali et al. (2009) indicated their distribution in a map (Fig.5.32). Hunting and culling data for the 2004–2005 season was at 34,027 individuals Carnevali et al. (2009).

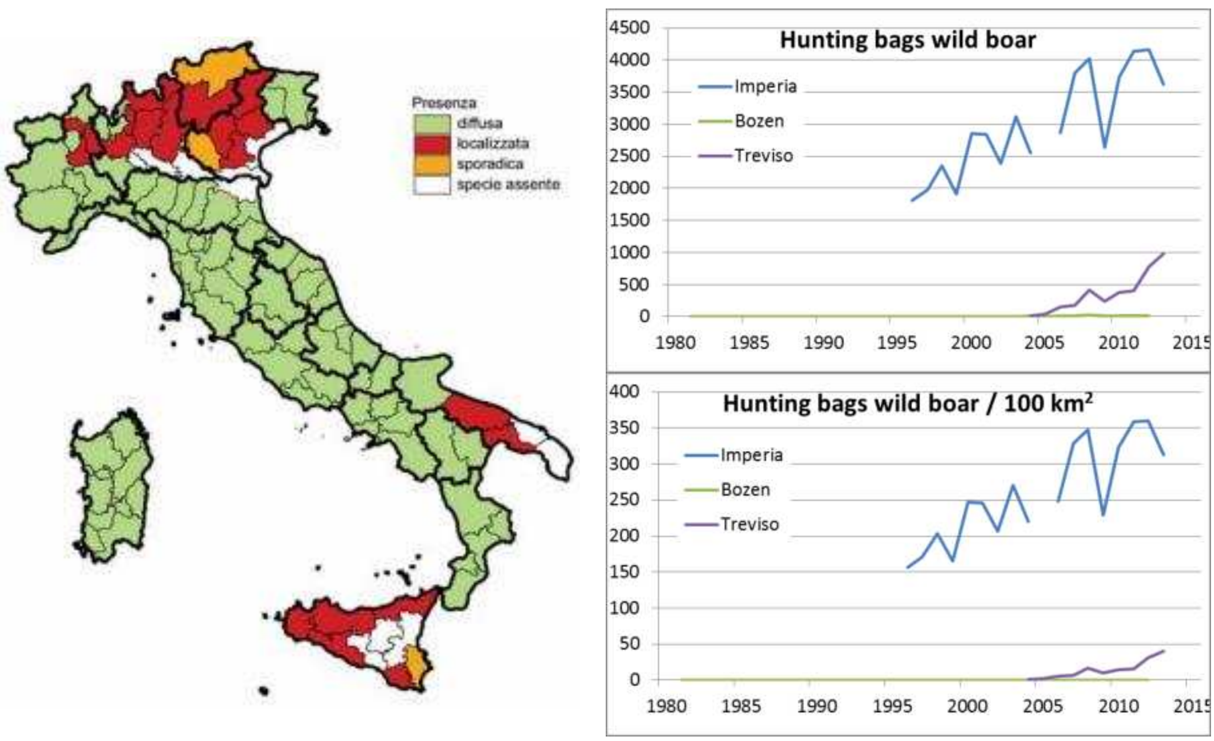


Fig. 5.32. Left: Presence of wild boar across the Italian provinces in 2005. Green= widespread, red= localised, orange= sporadic, white= absent (Carnevali et al. 2009). Top right: Hunting bags wild boar in three Italian provinces. Bottom right: Hunting bags wild boar per 100 km^2 in three Italian provinces (G. Torello, pers. comm., M. Stadler, pers. comm., S. Busatta, pers. comm.).

Switzerland. Wild boar populations in Switzerland recovered following natural immigrations from Germany, France and Italy during the second half of the 20th century (Imesch-Bebié et al. 2010). The wild boar population is fluctuating in Switzerland, which is also represented in the hunting bag statistics (Fig. 5.33): 9,940 individuals were hunted in 2012 and 5,740 in 2013 (BAFU 2014).

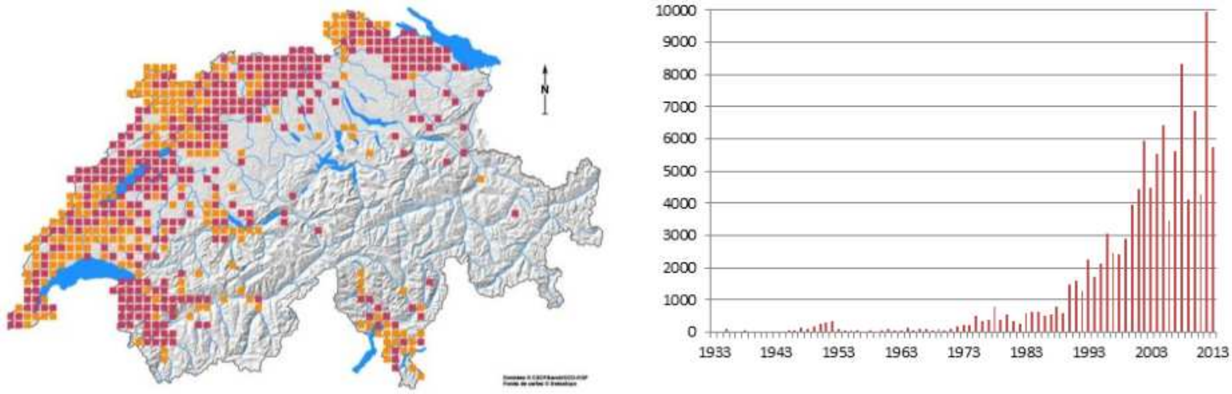


Fig. 5.33. Left: Distribution of wild boar in Switzerland (CSCF 2014). Orange squares: confirmed presence up to up to the year 2000, purple squares: confirmed presence since 2000. Right: Evolution of the hunting bag of wild boar in Switzerland (BAFU 2014).

Liechtenstein. There is no wild boar population in Liechtenstein. Dispersers occur from time to time, but they are hunted and cannot establish a population (W. Kersting, Amt für Umwelt, Liechtenstein, pers. comm.).

Germany. Wild boars were considered to be a pest species in Germany and were persecuted until the late 1950s (Wotschikowsky 2010). Despite occasional decreases due to environmental conditions, the wild boar population has increased steadily over the years (Fig. 5.34). However, it seems that the Alps (hunting grounds furthest south bordering Austria) are only slowly colonized.

Austria. Wild boar populations are thought to be increasing in Austria based on evidence of increasing culling rate (Fig 5.35; Reimoser & Reimoser 2010). An increase in wild-boar traffic kills is also a potential indicator of an increase in wild boar populations (Reimoser & Reimoser 2010).

Slovenia. In the beginning of the twentieth century, wild boar populations were low in Slovenia (Adamic & Jerina 2010). Wild boar populations have been low in Slovenia with populations occupying about 55% of the country of a potential 67% of available and suitable habitat (Jerina 2006). However, wild boar is still increasing in numbers and spatial distribution in Slovenia and has nearly tripled in the last 20 years (M. Jonozovič, Slovenia Forest Service, pers.comm.). Hunting bags in the three Alpine Hunting Management Districts averaged over the last ten years 738 individuals (Min: 513 in 2006; Max: 1,088 in 2008; M. Jonozovič, pers. comm.). Annual harvest rates have been found to be lower than in neighbouring countries (Adamic & Jerina 2010).

The evolution of hunting bag numbers for wild boar have increased across the Alpine countries and are fluctuating everywhere due to irregular mast years and hard winters (Fig. 5.36).

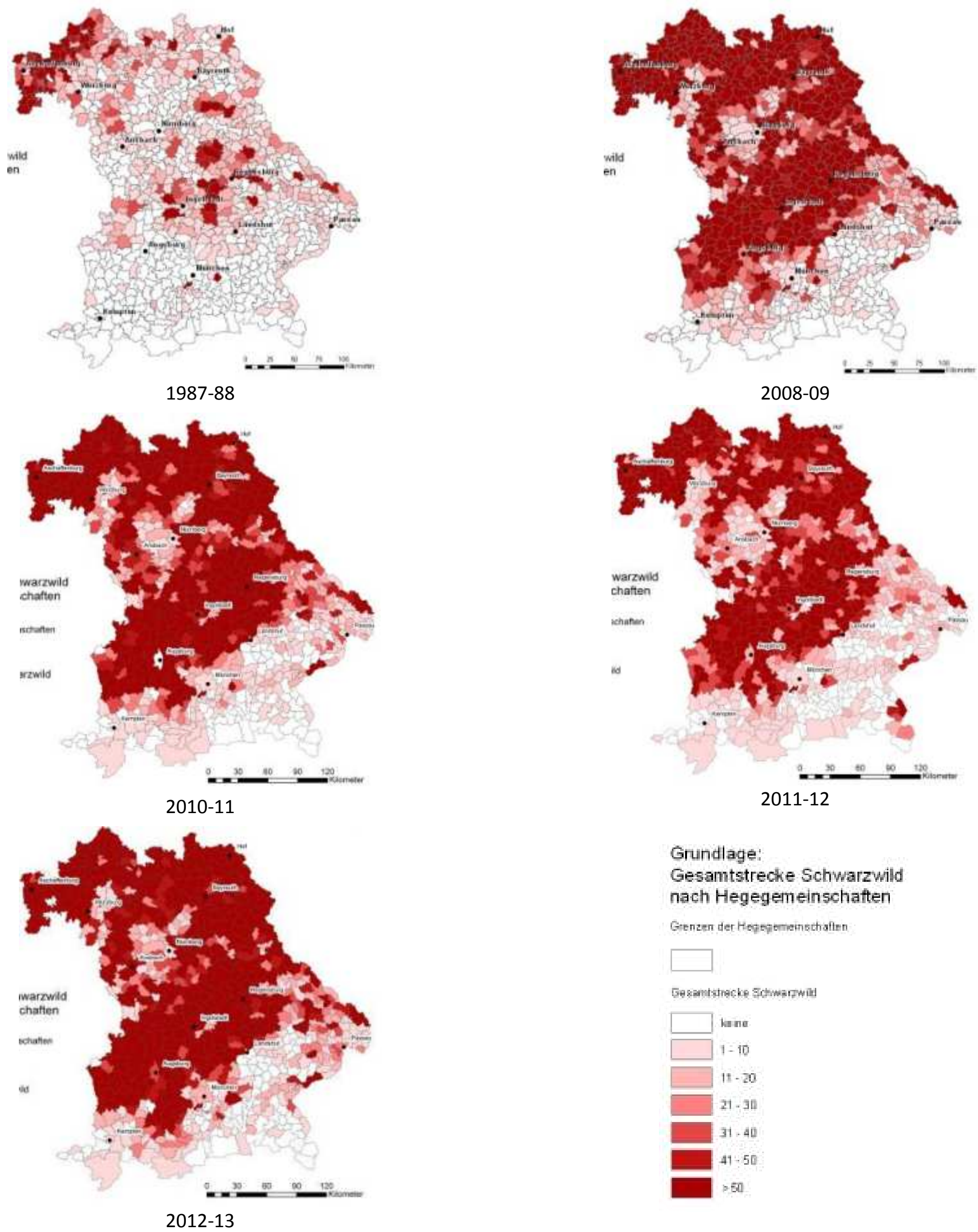


Fig. 5.34. Evolution of the distribution of wild boar in Bavaria, Germany, based on the hunting bag in the individual hunting grounds (Amtliche Statistik des Bayerischen Staatsministreriums für Ernährung, Landwirtschaft und Forsten 2014). Number besides the different shades in the legend represent number of hunted animals.

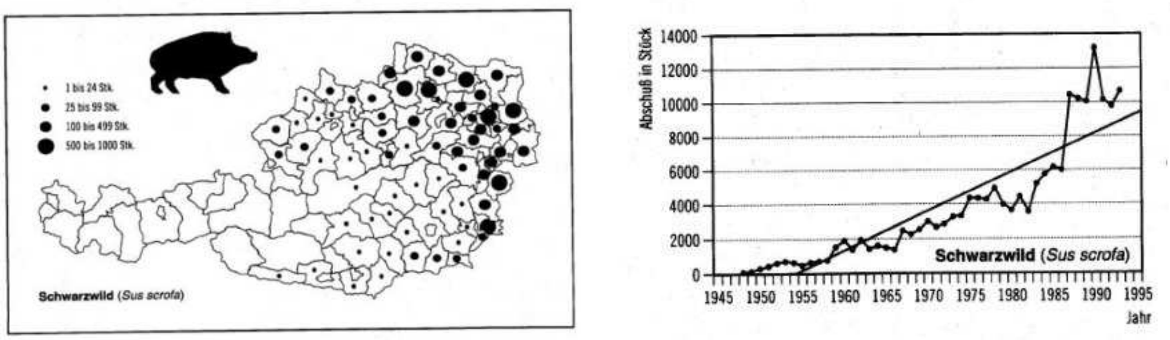
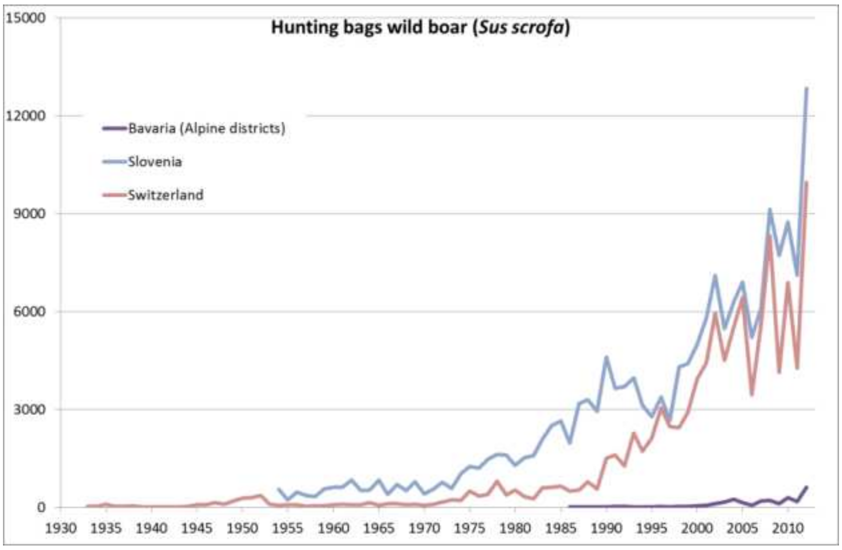
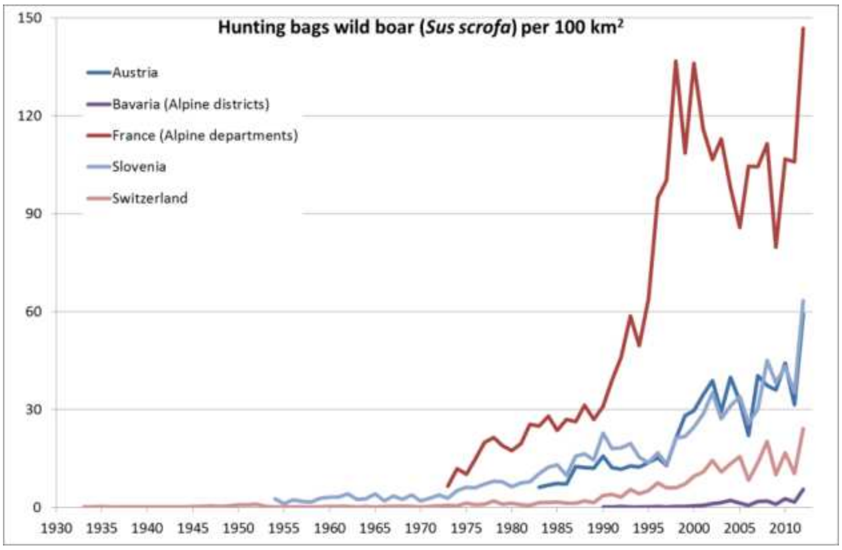


Fig. 5.35. Left: Wild boar hunting bag on district level (data from the average of the years 1986-1993). Right: Development of the wild boar hunting bag in Austria (1945-1995; Zeiler 1996).

a) Hunting bags for wild boar. Data for whole countries if not indicated otherwise



b) Hunting bags for wild boar (without Austria and France). Data for whole countries if not indicated otherwise



c) Hunting bags for wild boar per 100 km². Data for whole countries if not indicated otherwise.

Fig. 5.36. Hunting bag statistics for wild boar. Sources: AT: until 1982: see Breitenmoser & Breitenmoser-Würsten 2008; 1983-2013: Statistik Austria 2014; FR: Data ´Réseau Ongulés Sauvages´, ONCFS/FNC/FDC; DE: Reinhard Menzel, Bayerisches Staatsministerium für Ernährung, Landwirtschaft und Forsten, pers. comm., Friedrich Pielok, Bayerisches Staatsministerium für Ernährung, Landwirtschaft und Forsten, pers. comm., Frank Tottewitz, Thünen-Institut für Waldökosysteme, pers. comm.; SL: Statistical Office of the Republic of Slovenia 2014a; CH: BAFU 2014.

5.4. Livestock

Sheep are the most important and most abundant domestic species falling prey to carnivores in the Alps (Kaczensky 1996). Although several domesticated species can occasionally be victims of attacks of large carnivores, sheep and goat are by far the most vulnerable ones. The development of sheep and goat populations varied greatly across the Alpine countries. In France, Italy and Germany, the sheep and goat flocks decreased over the years, while populations in Liechtenstein and Austria showed an increasing trend (FAO 2014, see below). In France in 2002, about 450,000 sheep were grazing in Alpine regions where wolves were known to be present (Duchamp et al. 2004).

Pastoral systems and practices vary depending on the country, traditions and type of terrain. There are three main types of mobile sheep herding in Germany: nomadic, transhumance (i.e. seasonal change of grazing areas) and the alp system (Luick 2008). In France, the duration livestock remain in open pastures in the mountains is dependent on the type of mountain (Anonymous 2010). Sheep is by far the most often attacked livestock species (Chapter 5.5) and the only one having the potential to influence wolf (and to a much lesser degree lynx) distribution and local abundance, but its availability fluctuates considerably over the seasons.

5.4.1. Sheep populations in the Alps

Long-term trends of livestock in the Alps

When the large carnivores disappeared from the Alps in the second half of the 19th century, their main prey was livestock, because wild ungulates were scarce or even regionally extinct (Breitenmoser 1998a; Chapter 5.3). Now, as lynx and wolf are coming back, the prey base is very different compared to the times of their fall. Wildlife populations have recovered (Chapter 5.3), whereas livestock numbers have generally decreased (Fig. 5.37) or their husbandry practices were altered as a consequence of the economisation of agriculture. An exception are sheep, which have over the past 150 years lost their economic importance and show a long-term decreasing trend, but experienced a more recent re-increase in the eastern Alps. There are a number of local or regional historic publications to support such general statements, but no compilation of the long-term development of livestock numbers in the Alps is available. However, the trends were almost everywhere the same, as they were driven by demographic, economic and technological developments that were similar in all Alpine countries. We present the long-term statistics for livestock populations in Switzerland (Fig. 5.37), assuming that the trends across the Alps were similar or even stronger, as the effects of the industrialisation (rural exodus and vanishing of peasants) were even stronger in the southern and eastern parts of the Alps (Chapter 5.1).

In the past 150 years, livestock populations have seen considerable changes. Cattle experienced an increase, but also a concentration; more cattle are in fewer hands than 150 years ago. Horses have been replaced by tractors and trucks. Sheep, who are the main victims of large carnivore attacks, steadily decreased until World War II, but re-increased in the subsequent 50 years. Indeed, the decline of sheep started already around 1830, when the domestic wool production lost its competitiveness to wool from abroad and cotton. The sheep population started to recover in the 1950s, because sheep farming – and especially pasturing in the Alps in summer – became heavily subsidised. Sheep husbandry is promoted to prevent that remote pastures in the Alps are grown over by forest. Finally, goats have totally lost their former economic significance in the mountains, mainly because the typical owners of goats, peasants and landless families, had moved to the industrial centres.

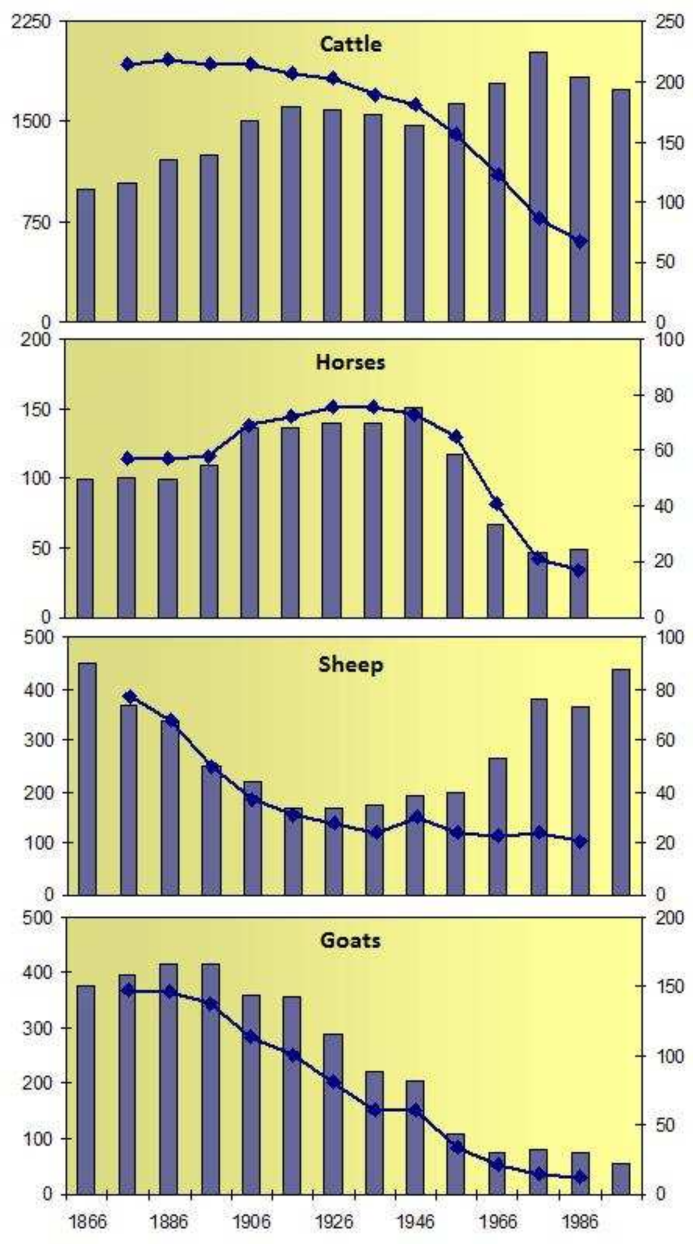


Fig. 5.37. Development of number of livestock (columns, left y-axes x 1000) and livestock owners (diamonds, right y-axes x 1000) in Switzerland since 1866 (beginning of national statistics; Federal Office for Statistics, www.statistik.admin.ch). The cattle population steadily increased, but the number of cattle farmers decreased. Horses were replaced by motorised vehicles and are today merely recreational animals. Sheep, the main domestic victim of large carnivore attacks, decreased until after World War II, then started again to increase to almost the same level as 150 years ago. Goats however have strongly decreased and lost their former significance in the Swiss Alps. (Sources: Breitenmoser 1998a, Breitenmoser & Breitenmoser-Würsten 2008).

Sheep populations in the countries of the Alpine Convention

Detailed data on livestock in the Alps are not readily available and national statistics are often difficult to break down into regions, and to interpret and compare between the countries, respectively. Numbers given in Fig. 5.38 refer to the whole countries sharing the Alps, as no such time series are available for the Alpine regions alone. Based on various data sets and the best possible geographic match with the area of the Alpine Convention, we estimated that the livestock populations in the Alpine area in 2010 were about 1.95 million sheep, 450,000 goats and 8.3 million cattle (Table 5.5). However, this includes all animals registered. The actual number of animals spending the summer on the Alpine pastures is only a fraction thereof. Of the 725,000 cattle registered in the Bavarian Alpine districts, only about 50,000 are summered on the pastures (StMELF 2010).

Table 5.5. Number of sheep, goat and cattle in the Alpine area. Data for 2010, except IT: 2007. Sources: FR: Insee 2014a, b; IT: ISTAT 2014; CH: Bundesamt für Statistik 2014; FL: Liechtensteinische Landesverwaltung 2014; DE: Bayerisches Landesamt für Statistik und Datenverarbeitung 2014; AT: Statistik Austria 2014; SL: Statistical Office of the Republic of Slovenia 2014b;.

	Data level	Sheep	Goat	Cattle
France	Département	833,142	100,042	421,787
Italy	Region	294,017	161,734	3,660,645
Switzerland	Canton	338,004	73,955	1,118,704
Liechtenstein	Country	3,656	416	5,993
Germany	District	36,389	19,583	725,057
Austria	Province	358,133	71,561	2,013,166
Slovenia	Statistical Region	88,868	26,101	363,173
Alpen		1,952,209	453,392	8,308,525

France. Alpine pastures in the French Alps belong to the Forest Department (Office National des Forêts ONF), the commune or to private owners (Biber 2010). Sheep owners rent the pastures from the respective land owner during the grazing season (Biber 2010). In France, shepherds often take sheep belonging to several owners and care for them together (Biber 2010). In general the numbers of sheep that are kept has decreased over the years (Fig. 5.38).

Italy. Like in some other countries, livestock husbandry decreased in recent years in Italy (Fig. 5.38).

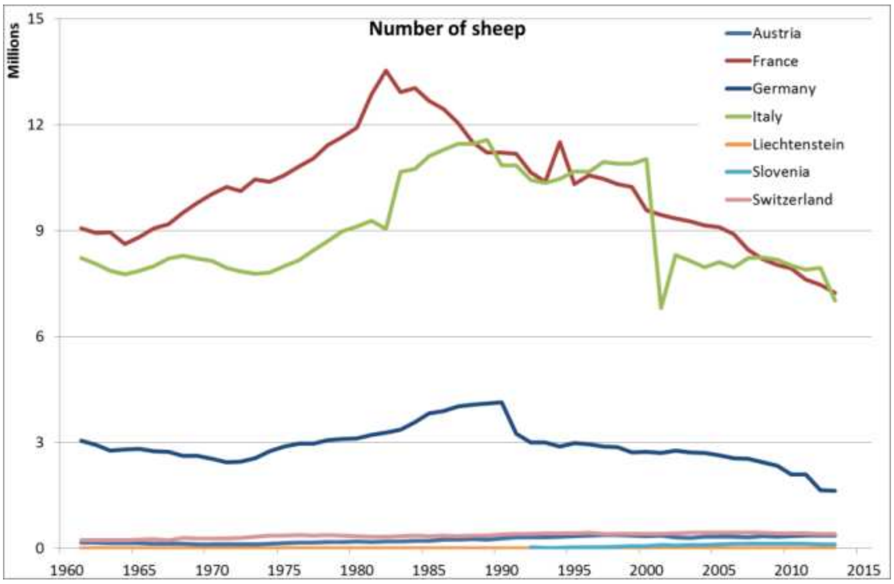
Switzerland. In Switzerland, sheep husbandry is no longer a lucrative activity and is therefore heavily subsidised by the government with the goal to prevent reforestation of remote pastures that are no longer used for grazing cattle (Breitenmoser 1998a). Sheep populations declined until the late 1940s and then started to increase followed by a decreasing trend in recent years (Fig. 5.38).

Liechtenstein. Unlike most Alpine countries, sheep populations have been rising steadily since the 1980s in Liechtenstein (Fig. 5.38).

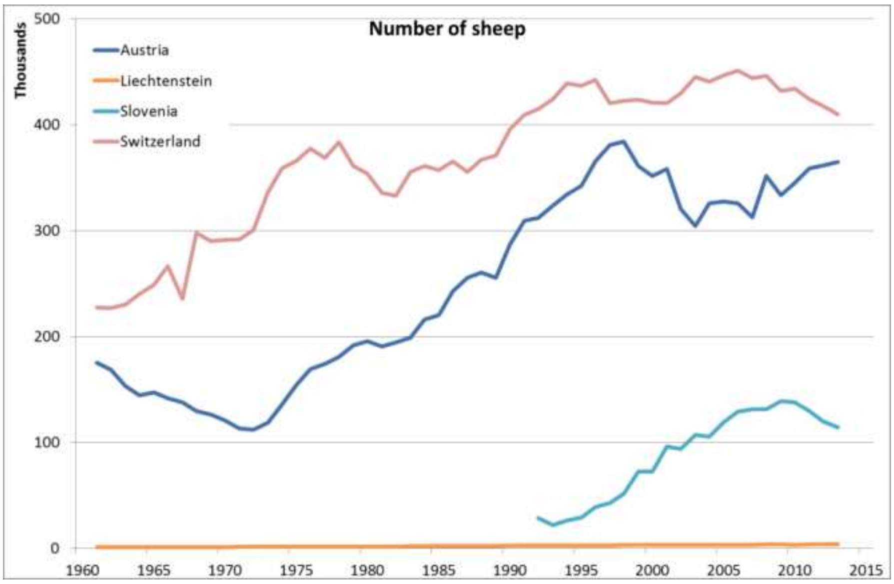
Germany. German livestock husbandry practises decreased at the end of the 1980s and owners kept fewer numbers of sheep over the years (Fig 5.38).

Austria. In Austria, like in Liechtenstein, the number of sheep herds is showing an increasing trend (Fig. 5.38).

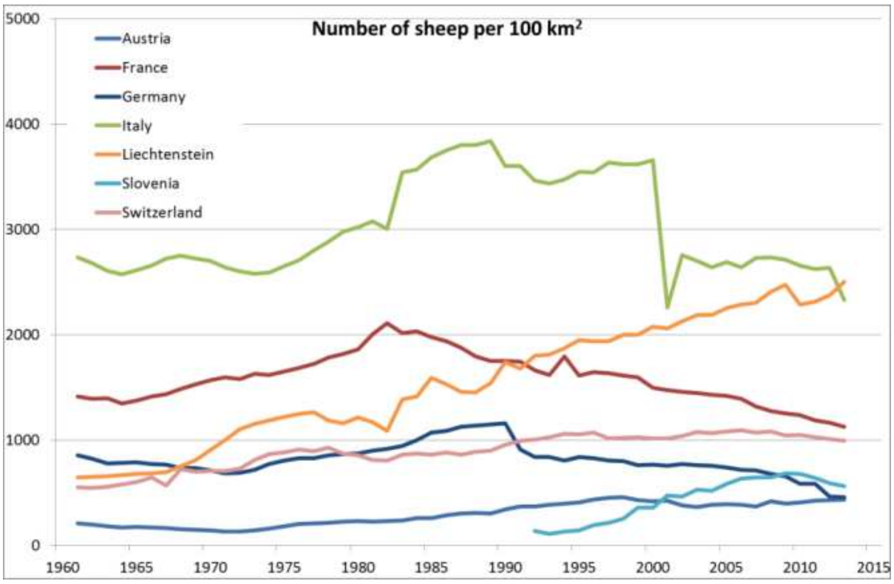
Slovenia. Since a few years, livestock population trend in Slovenia has been decreasing slightly (Fig. 5.38).



a) Number of sheep. Data for whole countries.



b) Number of sheep. Data for whole countries.



c) Number of sheep per 100 km^2. Data for whole countries.

Fig. 5.38. Number of sheep for the Alpine countries (FAO 2014, data for whole countries).

5.5. Predation: Diet of lynx and wolves in the Alps and impact on prey populations

Predation of large carnivores and their impact on wild prey populations and the losses they cause among domestic animals are the most important reason for conflicts with land users.

5.5.1. Predation of lynx on wild and domestic animals in the Alps

Generalities of lynx predation

Diet. Over 30 species have been recorded as prey species of Eurasian lynx. Their diet varies according to the geographic region (Odden et al. 2006). However, the main preys of lynx in Europe are small to medium-sized ungulates (Nowicki 1997), which are more or less of the same size as the predator. Wherever roe deer are abundant, they form the staple food, followed by chamois in the Alps and reindeer in the northern countries. Secondary prey species include red deer (fawns), foxes, hares, marmots, and (mainly in Scandinavia) tetraonids (Nowicki 1997, Breitenmoser & Breitenmoser-Würsten 2008). Ibex are very rarely preyed on and adult wild boars are never taken (Molinari & Molinari-Jobin 2003). Among domestic animals, lynx also concentrate on the small ruminants, namely sheep and goats. But the kill rate of domestic animals is on average low (see below) and lynx do not make surplus killings. An explanation for the preference for small ungulates was given by Jędrzejewski et al. (1993) based on body size, the solitary nature of lynx and avoidance of competition with other predators and pressure from scavengers. Indeed, roe deer and chamois are the only prey species in the Alps where lynx can potentially have an effect on their population. For all other prey species recorded, predation is too rare to have any impact.

Hunting tactics and handling of kills. A study on roe deer kills in Norway showed that there was no selection of roe deer based on age, sex or body condition (Andersen et al. 2007). This is mainly because lynx are ambush hunters, which make a surprise attack and do not “test” their prey for physical condition. They are more likely to kill animals that are not vigilant and are in situations where they can be easily approached. Lynx kill their prey through a bite to the throat either from below (larger animals) or from above (smaller animals) on the neck (Krofel et al. 2009). Lynx start eating larger prey from the hind quarters and in rare instances, from the shoulder. They consume all the muscle flesh and the heart, lung, liver, and kidneys, but leave the stomach and intestines untouched. Generally the head, skin and legs are all that remains (Capt 1992). The unmistakable features of a lynx kill are the hide turned inside-out over the head and the carcass camouflaged with grass, leaves, or snow (Molinari et al. 2000). Lynx may drag a large kill into the closest cover available, but generally, they leave the prey where it was killed and return to it at dawn. Females bring the cubs to the kill and only exceptionally carry food to the kittens (Breitenmoser & Breitenmoser-Würsten 2008).

Kill rate. Depending on the size of the prey, lynx can return from one to seven consecutive nights to feed on their kill (Capt et al. 1993, Breitenmoser & Breitenmoser-Würsten 2008). A large chamois was even reported to be consumed by a single lynx over 12–15 days (Herrenschmidt & Vandel 1989). Adult chamois were reported to be consumed during 4.6 days, while male roe deer lasted 4.3 days and female roe deer lasted 4.0 days (Jobin et al. 2000). Domestic sheep or goat kills are often abandoned before they are completely consumed, because these cadavers are removed or the lynx is shooed away by the presence of people. Average daily consumption of meat per day was estimated to be 2 kg with a maximum of 3.0–3.5 kg (Haglund 1966, Bufka & Cerveny 1996). Small kills such as fawns, hares and birds can be consumed in one sitting. The yearly consumption rate for lynx in Switzerland, based on radio-telemetry studies (Breitenmoser & Haller 1987, Haller 1992, Jobin et al. 2000, Molinari-Jobin et al. 2002), was estimated to be 56 ungulates (roe deer and chamois) for an adult male lynx, 57 for subadult lynx, 59 for solitary females, and 72 for females with cubs. So considering the share of the social categories in the lynx population, this would come to about 61 ungulates per independent lynx and per year (Breitenmoser & Breitenmoser-Würsten 2008).

Predation studies in the Alps

France. Main prey species of lynx in the French Alps are roe deer and chamois. Herrenschmidt & Vandel (1990, 1992) reported roe deer as being an important prey species with a selection for females over the age of seven years. Other wild prey species included foxes and hares.

In the French Alps, depredation rates were low, presumably due to the low lynx population as well as the use of protective measures in some regions (Chapter 4 and 6.3.1). Moreover, domestic livestock are never the main prey (Stahl et al. 2001). Only five cases of depredation were reported in 2010 in the department of Rhône which is outside the Alpine area (ONCFS 2011). The average number of depredation cases attributed to lynx for the whole of France between 2000 and 2011 was 72 (Marboutin 2013b). However, most of these cases occurred in the Jura Mountains.

Italy. The lynx presence in the eastern Italian Alps, the Tarvisiano, was studied by Molinari (1998). Species that were preyed on during the period between 1987 and 1995 include roe deer, chamois, red deer, hare, marmot, capercaillie and sheep. The study showed that the lynx readily predated on red deer due to its high abundance.

Switzerland. Long-term studies using radio-telemetry showed that 90% of the lynx diet is comprised of roe deer and chamois (Breitenmoser & Haller 1993, Molinari-Jobin et al. 2007, Breitenmoser & Breitenmoser-Würsten 2008, Breitenmoser et al. 2010). Although lynx diet was found to vary in different regions of Switzerland (Fig. 5.39; Breitenmoser & Breitenmoser-Würsten 2008), the main prey is roe deer, followed by chamois (Fig. 5.39a–c). In the central Alps after the first colonisation phase of lynx, chamois was temporarily the most important prey (Fig. 5.39e; Haller 1992, Breitenmoser & Haller 1993) and seemed to be selected for especially in areas with high abundance (Molinari-Jobin et al. 2007). Female lynx in the north-western Alps preyed mostly on roe deer, while male lynx, being about 30% heavier, kill significantly more grown-up chamois (Breitenmoser & Breitenmoser-Würsten 2008).

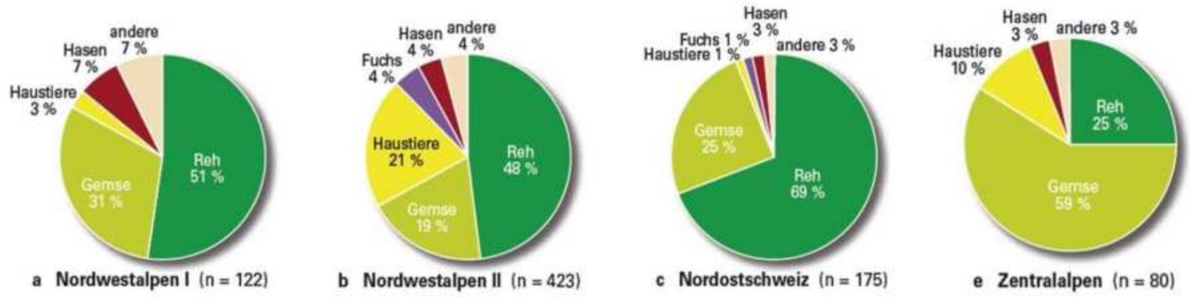


Fig. 5.39. Lynx prey from different studies based on radio-telemetry in the Swiss Alps (Breitenmoser & Breitenmoser-Würsten 2008). The regions include (a) the north-western Alps I (1983–1988) and (b) II (1997–2001), (c) north-eastern Switzerland (2001–2003) and (e) central Alps (1985–1988). Species shown include roe deer (green), chamois (olive), domestic animals (yellow), hare (red), red fox (violet), and others (pink).

Domestic livestock are preyed on to a certain extent. Between 1973 and 2013, 2052 sheep and 219 goats were confirmed to have been predated on by lynx in the whole of Switzerland and compensated by the state (other= 81, total= 2352; Fig. 5.40; Kora, unpubl. data). Attacks occurred wherever unguarded sheep and goat are available within the lynx range (Fig. 5.41), though on low level. Livestock depredation by lynx was analyzed for the period of 1973-1999. In the peak year 1999, the loss of sheep was estimated to be no more than 0.4% of the total stock, and two thirds of the flocks were never attacked. Only 22% of the flocks were visited in more than one year. Nevertheless some flocks were preyed on more often than others and suffered considerable losses (Angst et al. 2000). Depredation occurred mostly between June and October when sheep are grazing in high altitude summer pastures, with occasional cases in March, April, May and November, based on depredation data between 1979 and 1999 (Angst et al. 2000).

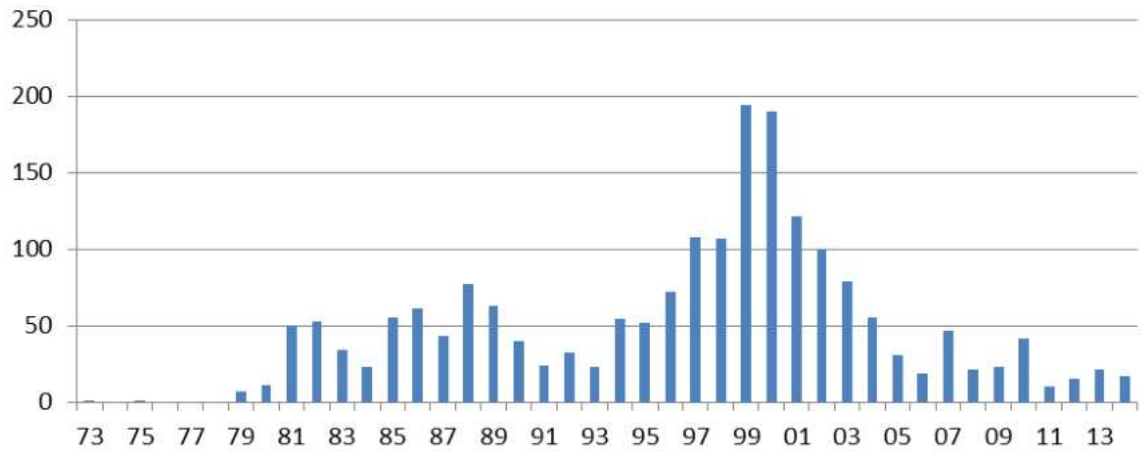


Fig. 5.40. Attacks of lynx on domestic animals (89% sheep, 8% goats, 3% others) in the Swiss Alps since reintroduction of the predator in 1971. There were several "peaks" of local attacks, and they were all correlated with a reduced availability of the main wild prey species, the roe deer. The most prominent peak occurred in the years 1999 and 2000 in the north-western Alps. As long as wild prey are abundant, lynx attacks on livestock occur only sporadically (KORA 2014).

The Swiss lynx management plan (Chapter 6.1) allows the removal of notorious stock raiders. Removing problem lynx proved to end the attacks in half of the cases (Fig. 5.41). In the other situations, however, lynx newly moving into the area after the former resident lynx was removed continued to prey on sheep in the same pastures (Angst & Breitenmoser 2003). These pastures were considered "hot spots", where sheep obviously were so exposed that any lynx would attack sheep. This might have been a consequence of the pasture (mainly bushy pastures in well-forested areas) or the local lack of wild prey. In these cases, only measures to protect the flock or the removal of the flock were able to stop the attacks.

Sheep flocks in Switzerland are only exceptionally actively protected against lynx attacks, as such attacks are too rare (and generally only one animal per night is killed). Hence the usual policy is to (1) compensate occasional losses without further measures, (2) remove notorious stock raiders, and (3) implement protective measures or remove the flock in special cases (e.g. more than one lynx attacking the same flock; Angst & Breitenmoser 2003). In recent years, wolf attacks on livestock have by far outweighed lynx attacks; and the presence of wolves has promoted the large-scale application of herd protection. In the Swiss Alps, periods of lynx depredation were related to low availability of wild prey. (For different experiences in the French Jura Mountains see Stahl et al. 2001). This is most likely due to the usually high abundance of wild prey and the natural tendency of lynx to prey on wild ungulates when available (U. Breitenmoser, pers. comm.). Between 2006 and 2011, only 7 to 47 sheep were predated on per year by lynx in the Swiss Alps (von Arx & Zimmermann 2013).

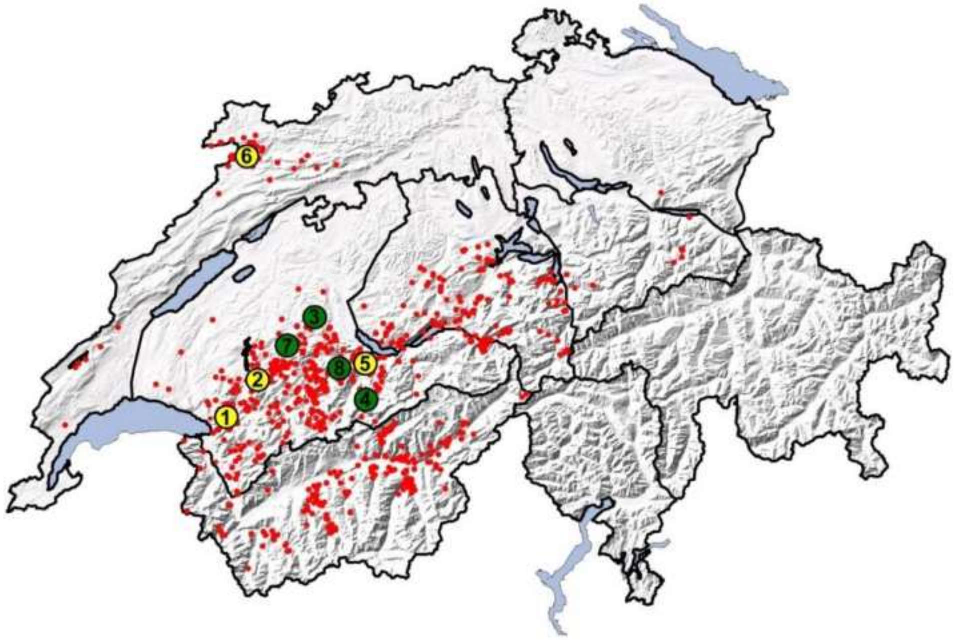


Fig. 5.41. Attacks on livestock by lynx in Switzerland 1993–2004 (red dots). Most attacks remained single cases. If a lynx notoriously kills livestock, the authorities can issue a permit to shoot it. 8 lynx were removed in the period shown here. In half of the cases (green dots) the attacks stopped after the lynx was shot, but in the other four cases (yellow dots), the attacks continued (Breitenmoser & Breitenmoser-Würsten 2008).

Liechtenstein. Although lynx currently do not have an established population in the country, dispersing individuals moved through the mountains of Liechtenstein (Fasel 2001, 2003). Some prey remains were found in the past few years, but it was difficult to attribute them to lynx (Frick 2012). Game wardens and hunters are encouraged to report kills and other signs which would indicate lynx presence (Frick 2012).

Germany. Livestock depredation is generally rare in Bavaria. On the one hand, there is no lynx population in the German parts of the Alps. On the other hand, sheep farming is not an important activity in the German part of the Alpine arc (Kaczensky et al. 2013b). Nevertheless, a number of unconfirmed kills were recorded in the German Alps between 2005 and 2009 (Wölfl & Wölfl 2011).

Austria. After its reintroduction in the mid-1970s, Gossow & Honsig-Erlenburg (1985) reported a preference for red deer in lynx diet in the eastern Austrian Alps in areas with a very high red deer abundance. Other potential prey included chamois, although they were on the fringe of their suitable habitats (Gossow & Honsig-Erlenburg 1985), whereas roe deer were not present in their study site. Newer anecdotal observations of newly released and radio-collared lynx in 2012–2014 revealed that roe deer seemed to be the main prey, too (C. Fuxjäger, pers. comm.). Nowicki (1997) compared the results of several studies examining lynx diet in various countries across Europe. The lynx diet studies in Austria showed a preference for red deer and roe deer (Fig. 5.42; Gossow & Honsig-Erlenburg 1983).

In the period following the reintroduction of the lynx in Styria, there were few cases of depredation reported mainly from Carinthia, but the diagnosis of lynx kills was often unclear. In 1987, 28 sheep were reported to have been killed by lynx in the southern part of the Koralpe (Huber & Kaczensky 1998). Compensations for sheep depredation were subsequently claimed over the following years until 1995 (Huber & Kaczensky 1998; Chapter 5.4). The greatest number of livestock killed was in 1989 when 48 sheep, 10 lambs, 1 calf and 1 goat were confirmed to have been predated on by lynx (Huber & Kaczensky 1998). An examination of several prey remains collected between 2005 and 2009 included two cases of livestock depredation and two fallow deer (Fuxjäger et al. 2012).

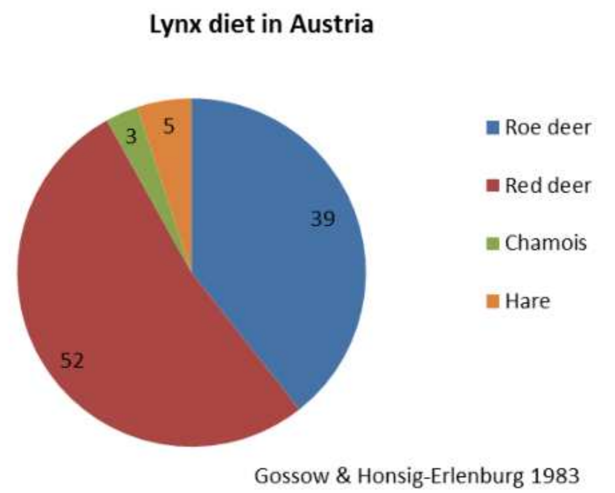


Fig. 5.42. Lynx diet in Austria based on scat analysis, data presented as percentage of biomass (Gossow & Honsig-Erlenburg 1983).

Slovenia. Lynx occur mainly in the southern part of the country, outside the Alps. Only few lynx were observed in the Slovenian Alps, but data are generally reported for the whole of the country. Lynx diet in the Dinaric Mountains of Slovenia was studied by Krofel (2006) who found that roe deer and red deer were primary prey species. Roe deer comprised 64% of the diet while red deer 24% of total biomass consumed (Krofel 2006). Animals selected for were in poor physical condition (Krofel 2006). Subsequent studies in the Dinaric Mountains in Slovenia and Croatia showed that roe deer represented 79% of all consumed biomass in the lynx diet (Krofel et al. 2011). Edible dormice were found to be a recurring prey consisting of 7% of the total biomass consumed (Fig. 5.43; Krofel et al. 2011).

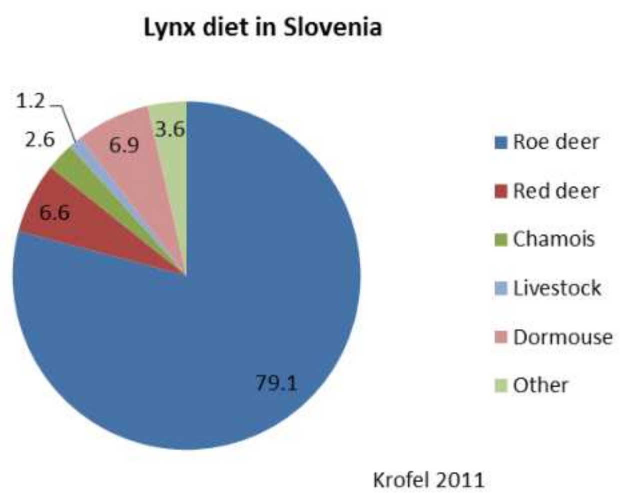


Fig. 5.43. Lynx diet in Slovenia based on percentage of biomass consumed. The category "Other" includes red fox and hare (Krofel et al. 2011).

Livestock depredation is low due to the low density of lynx and availability of wild prey (Kaczensky et al. 2013d). A total of 317 depredation cases have occurred in the Alps since 1994. In 2001, a record of 88 cases occurred, which is more than double the number of cases in any other year. No livestock depredation by lynx has been recorded since 2011 (M. Jonozovič, pers. comm.).

Impact of lynx on prey populations

Considering an average yearly consumption of 60 small ungulates per year and lynx (see above) and a mean long-term lynx density of 2.0 independent lynx/100 km² (resident adults and dispersing subadults; Chapter 7; Appendix III), the toll of lynx predation on prey populations would be moderate. A 100 km² area in the north-western Alps would host some 400 roe deer and 220 chamois (calculated based on official wildlife estimations of the canton of Bern; BAFU 2014) and a loss of about 72 roe deer and 48 chamois (ratio according to Fig. 5.39a) to lynx predation would be long-term sustainable. However, such "long-term averages" do not reflect the possible predation impact in a local and temporary situation. Under certain conditions, depending e.g. on the population status of predator and prey, on the recolonisation state (e.g. immigrating predators), and on other important mortality factors (e.g. winter mortality or human-made mortality), lynx can have a significant impact on a local roe deer population. This has e.g. been described by Samelius et al. (2013) for central Sweden, and Okarma et al. (1997) for Białowieza in eastern Poland.

Another effect of lynx presence and a potential source of conflicts with hunters are behavioural changes in prey species, especially increased vigilance that make hunting more difficult. Although this has been claimed repeatedly and seems to be correlated with high lynx densities, it has never been addressed by a scientific project and hence there is no robust data to assess this aspect.

The longest observation series to address the quantitative impact of lynx predation for the Alps are available for Switzerland (Breitenmoser et al. 2010). The observed roe deer mortality caused by lynx predation (based on the assumed local abundance) varied from 9 to 63% (Breitenmoser et al. 2007) indicating how strong the predation impact can change. Indeed, the extreme values (9 and 63%, respectively) came from the same study area in the north-western Alps about 15 years apart. Haller (1992) observed a strong predation on roe deer in a high valley in the central-western Alps (Canton of Valais) shortly after the reintroduction of lynx. A sudden drop of the roe deer population was also observed in Central Switzerland (Canton of Obwalden) about 10 years after lynx had been reintroduced (Breitenmoser & Breitenmoser-Würsten 2008). Such observations were attributed to the continued lack of vigilance and behavioural adaptation of roe deer after the recolonisation by lynx (Breitenmoser & Haller 1993). However, another period of strong predation impact was observed in the years 1997–2000 in the north-western Alps, where lynx had been present for almost 30 years. This case – although only a case study – illustrates the potential predation impact and the cofactors that influence the predator-prey system and should be considered in a wildlife management system in the Alps with large carnivores present.

In the late 1980s and early 1990s, probably as a consequence of a series of mild winters and low winter mortality, the roe deer population in the north-western Alps increased, triggering a numeric response (an increase with a certain time delay) in lynx (Fig. 5.44a). The growing roe deer population had also triggered an increasing human hunting pressure (Fig. 5.44b), demanded for by the foresters. After about 1995, the roe deer population started to decline, first slowly and then faster, causing a reduced hunting bag, as the hunters were no longer able to fulfil the quota. During this time, the lynx population still increased, and the predation impact reached its peak in the years 1997–2000. Lynx were in these years responsible for about 60% of the know mortality in roe deer and about 33% in chamois. During the period of low roe deer abundance, lynx predation was estimated to affect 36–39% of the estimated spring population of roe deer every year (Breitenmoser & Breitenmoser-Würsten 2008). Lynx maintained a rather high predation pressure on roe deer in spite of (or as a consequence of) part-switching to other prey, namely to chamois and more obviously to sheep (Fig. 5.44a).

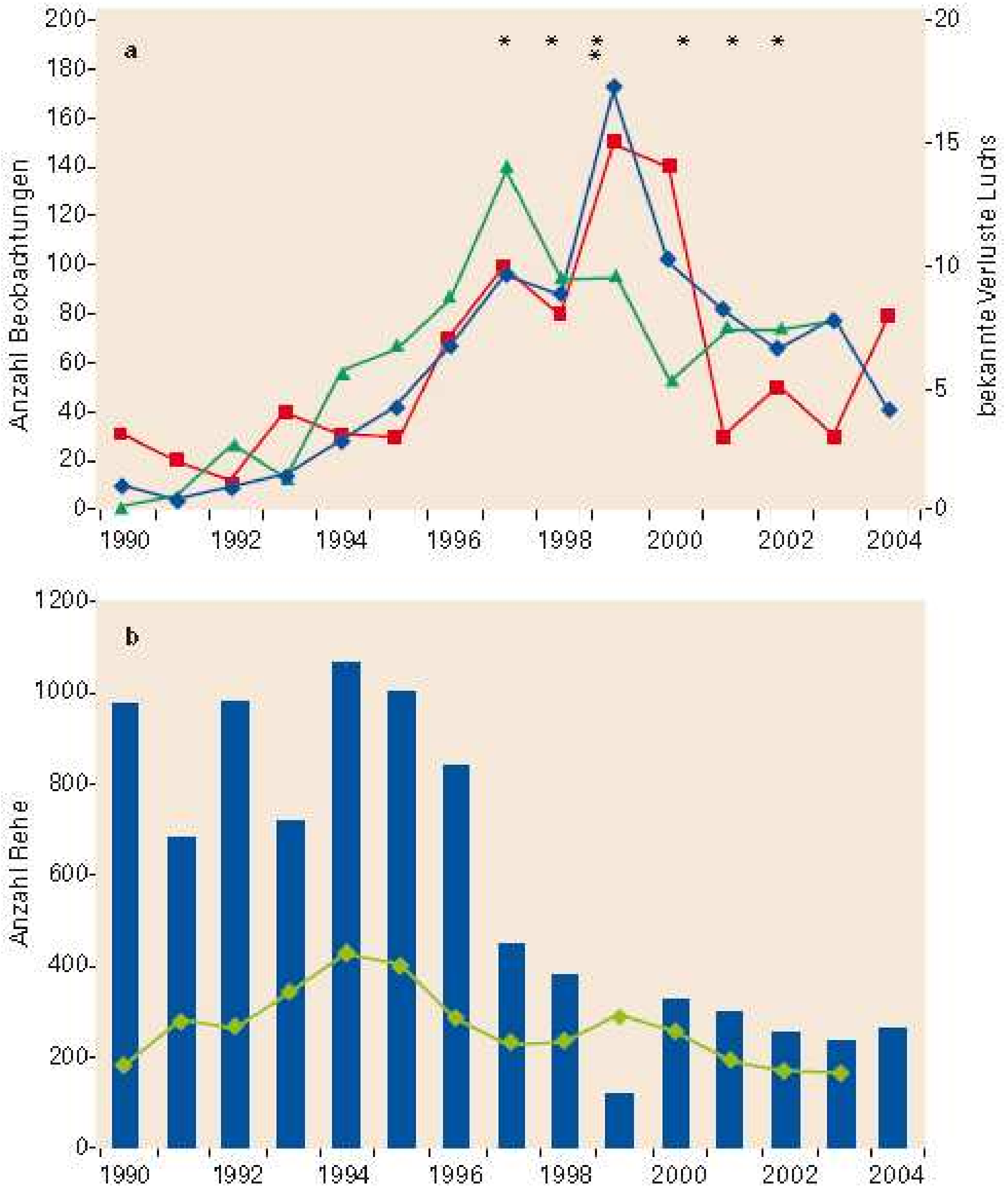


Fig. 5.44. Development of the lynx population (a) and the known roe deer mortality (b) in the north-western Swiss Alps 1990–2004. After 1992, the number of chance observations of lynx (a, green curve, left y-axes), the known lynx mortality (a, red curve, right y-axes) and the number of attacks on livestock (a, blue curve, left y-axes) steadily increased, indicating a growing lynx population. In the late 1980s and early 1990s, the hunting bag of roe deer also increased (b, histogram), parallel to a slight increase of the number of roe deer killed in traffic accidents in the same area (b, green curve). After about 1996, a drastic decline in the roe deer population was observed, illustrated by a drop of the hunting bag and a decreasing number of traffic victims. After a "lynx peak" in 2000, the lynx population dropped, too. In these years, 8 lynx (a, asterisks) were removed as stock raiders (source: Breitenmoser & Breitenmoser-Würsten 2008).

This example illustrates how strong lynx predation can be under certain conditions and in this case supported by anthropogenic and climatic factors. Although difficult to demonstrate, the roe deer increase and subsequent fall was supported by a series of mild winters in the early and harsh winters in the late 1990s. Furthermore, the predation impact of lynx and the hunting pressure from human hunters were increasing parallel in the early 1990s, up to the moment when the combined effect of lynx predation, hunting, and winter mortality caused a swift decrease of roe deer abundance. After the peak years, lynx showed a negative numeric reaction, but we cannot estimate its importance, as it was strongly supported by “management measures”, including the lethal removal of stock raiders, the translocation of 9 lynx in the years 2001–2003 into the eastern Swiss Alps, and a clear increase in illegal killings of lynx (Breitenmoser & Breitenmoser-Würsten 2008). In winter 1998/99, the local abundance of lynx was estimated 2.6 independent individuals/100 km², and it dropped to 1.0 lynx/100 km² in winter 2001/02 (based on capture-mark-recapture estimations based on camera trapping; Laass 1999, 2002).

Such a predation impact is exceptional, but it illustrates the potential and the management challenges. In the Alps, winter mortality is essential for the population dynamics of all ungulates, but especially for the small roe deer. On the one hand, the habitat e.g. of the northern Alps can support a high roe deer density, which can be reached when winters are mild. On the other hand, a single harsh winter can cause a considerable population decline. If such a situation coincides with a numeric response by lynx and with management measures (e.g. an increased hunting quota to satisfy the demands of foresters), it can lead to a locally and temporarily very high predation impact.

Predation on chamois, the second-most important prey species of lynx in the Alps, has so far not been demonstrated to be as significant for the population as on roe deer, although in the above mentioned case study, chamois predation was also high as a consequence of a partial switching to chamois when roe deer availability was low.

Chamois are hunted by lynx mainly if they stay in the forest. Molinari-Jobin et al. (2004) revealed for the Jura Mountains, that chamois in forested areas are indeed more vulnerable to lynx predation than roe deer. However, a good part of the chamois population is staying above the timberline, where lynx hunt less. But recent observations indicate that chamois may be more affected by predation on juveniles than roe deer (KORA, unpubl. data, Fig. 5.45).

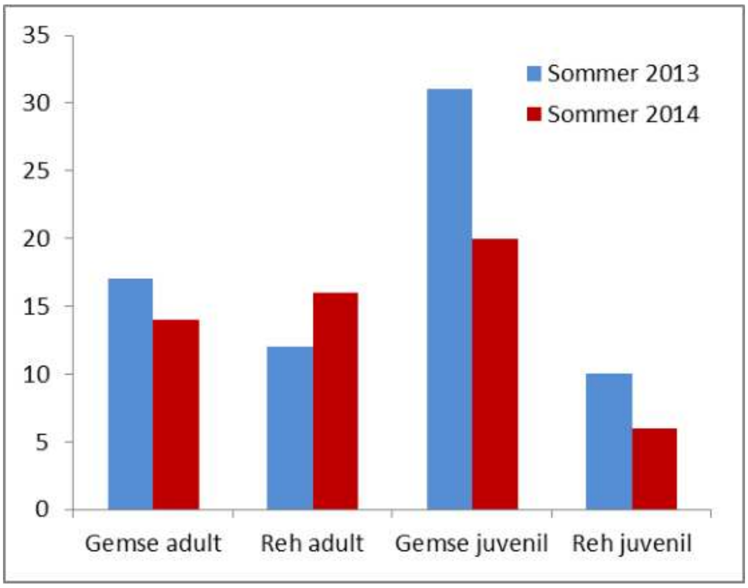


Fig. 5.45. Predation of lynx on young and adult roe deer (Reh) and chamois (Gemse) in the north-western Swiss Alps, based on a GPS-telemetry study (KORA, unpubl. data).

The breeding season of lynx, roe deer and chamois is rather well synchronised. We can assume that the predators adjust the time of giving birth to the season when new-born prey is readily available and that they should switch to new-born and young prey animals during the summer months. Preliminary observations based on GPS telemetry suggest that indeed juvenile chamois are preferably taken, but not so juvenile roe deer (Fig. 5.45). Whereas in the summer months, juvenile chamois were taken more than adults, and significantly more than roe deer fawns, juvenile roe deer, although they are very abundant in these months due to the high reproductive output of the species, were less often killed by lynx than the other prey categories. This may be because roe deer fawns hide, whereas juvenile chamois are followers and therefore more obvious. Considering the relatively low reproductive capacity of chamois, a strong predation on juveniles could have an impact on the demography of the species. But the data presented above are so far too preliminary to draw conclusions at the level of the population.

Although lynx have large exclusive home ranges and occur at low densities (Chapters 4.2, 7; Appendix III), they have the potential to show a distinct numeric response to increased prey availability. Local lynx densities can fluctuate by a factor 2.5–3, what is considerable for a large carnivore in the temperate zones. If peak lynx densities meet decreasing or low roe deer densities, the predation impact can be strong. Such a situation will not last, as lynx density will subsequently decrease, too. However, in each such situations so far experienced in the Swiss Alps, the immediate consequences were severe conflicts with hunters. Part of the story is (unpredictable) effects of winters and winter mortalities. However, it is important to understand the ecological features and population dynamics of predation and predator-prey relations to anticipate changes in predator and prey densities and hence conflict with human activities such as hunting or sheep husbandry.

5.5.2. Predation of wolves on wild and domestic animals in the Alps

Generalities of wolf predation

"[T]he diet of the wolf is as broad as its geographic range" (Peterson & Ciucci 2003). This statement underlines the ubiquity of wolves and their omnivorous character. Understanding how a species feeds itself is an integral part of understanding its behaviour and therefore a prerequisite to developing appropriate and focused management actions especially in the case of wolves, with regard to the importance of livestock in their diet. Wild and domestic ungulate populations are determining elements of wolf presence, abundance, pack size and movements. Prey abundance and large ungulate biomass are related to the number of young wolves in autumn and general wolf abundance in an area (Fuller 1989, Fuller & Sievert 2001). Dependence on domestic prey is related to wolf density, home range sizes, availability of wild prey, seasonality and livestock protection practises (Fluhr 2011; Chapter 6.3.3).

Scat analysis is a popular method to study the diet of wolves; however, interpreting results obtained using this method requires caution due to sampling errors and biases (Marrucco et al. 2008, Milanesi et al. 2012). Alternative methods consist of the analysis of stomach contents of dead animals (Zunna et al. 2009), or searching directly for kills either by snow-tracking (Jędrzejewska et al. 1994) or by following radio-collared animals (Sand et al. 2008), noting the prey species and the completeness of its consumption. The different methods can also be used in combination (Palmegiani et al. 2013). Wolves are also scavengers; the occurrence of a prey species in wolves’ stomachs and/or scats does hence not necessarily need to come from animals that were actually hunted and killed. On the other hand, ravens, brown bears, coyotes, foxes and other species also often scavenge wolf kills. Therefore, even though a kill may not be fully consumed, it is of importance to other species (Mech & Boitani 2003, Peterson & Ciucci 2003).

Wolves are highly adaptable and can travel up to 45 km in a single night in search of prey but can also survive on garbage and waste materials in human dominated landscapes (Zimen & Boitani 1975, Peterson & Ciucci 2003). They can also survive long bouts of low food availability (Peterson & Ciucci 2003). Fruit can constitute a significant part of the diet when conventional prey is scarce (Meriggi et al. 1991). The omnivorous character, the ability to scavenge, to travel far and to take advantage of clustered food sources has historically enabled wolves to survive in areas where wild ungulates became scarce and lynx (who depend on wild prey they kill themselves; Chapter 5.5.1) have went extinct.

Prey choice in Europe is seasonal and is mainly dependent on the availability of vulnerable individuals among wild ungulates. “Vulnerability” is not restricted to physical weakness, but also to a question of the social status and behavioural constraints. E.g. subadult or old animals living outside the social groups, or males during the rutting season are more vulnerable as they are less attentive to their surroundings, but also individuals in poor physical condition (disease, weight, parasites, injuries and abnormalities, nutritional condition) and young animals are often selected (Nelson & Mech 1986a, Mech & Peterson 2003). A seasonal vulnerability in ungulates leads to an especially high predation rate during winters with deep snow cover (Nelson & Mech 1986b, Espuno 2004).

Diet. Numerous studies in Europe found that wolves in general preferred to prey on wild ungulates, especially cervids (Bassi et al. 2012). The abundance of red deer and the strong positive response of wolves to red deer density determine the proportions of other species in the wolves’ diet (Jędrzejewski et al. 2000). A review of 20 studies performed in Italy between 1976 and 2004 found a general positive correlation on a national and regional level between the abundance of wild ungulates and their frequency of occurrence in the diet of wolves (Meriggi et al. 2011). Milanesi et al. (2012) performed their study in the same area as Meriggi et al. (1991) and compared the results. Significant increases in the abundance of wild ungulates led to a significant increase of wild ungulates in the wolves’ diet too. At the same time, livestock depredation decreased from 41 heads killed in 1991 to 10 and 20 heads killed in 2007 and 2008 respectively, despite an increase in the number of packs from two to four (Milanesi et al. 2012).

Another aspect of the adaptability of wolves is shown in the seasonal variation in their diet. For example, cervids are the primary prey of wolves in the Italian Alps both in summer and winter. However, the relative contribution to the diet decreased in summer and while the secondary prey in winter consisted of chamois, livestock became more important in summer (Gazzola et al. 2007). Wolves in Germany were found to prey frequently on juveniles, females and old animals (Wagner et al. 2012). In the Italian Alps, juvenile red deer and juvenile roe deer constituted an important portion of wolf diet (Gazzola et al. 2005). The number of males in the diet increased during the rut as they were less attentive to their surroundings, are physically stressed and therefore more vulnerable to attacks from predators (Mech & Peterson 2003, Palmegiani et al. 2013). Shifts in ungulate populations also lead to shifts within the wolf diet and prey choice; the decrease in mouflon populations in the southern French Alps led to wolves shifting from a diet based mainly on mouflon to other species such as the roe deer and wild boar (Fluhr 2011). In cases where prey densities have changed or been altered (changing livestock protection regimes), wolf predation patterns have changed in a way that they choose the prey that is most easily available (Garrott et al. 2007). Predation on livestock can be linked to the wolf's biology: during the birthing season, they require a higher nutrition and livestock are an easy source of prey requiring low energy to obtain (Mattielo et al. 2012).

Hunting tactics and handling of kills. Wolves tend to live and hunt in packs. There are various theories regarding the relationship between pack size and prey size. Pack size can also influence the importance of domestic prey in the diet; smaller packs have a greater tendency to prey on livestock compared to large packs (Fluhr 2011). However, in North America, single wolves have been known to bring down prey as large as adult moose (Mech & Boitani 2003). In Europe, pack sizes tend to be small as wolves are still in the process of recolonizing their historical range. Pack size and hunting group size are not necessarily synonymous: in winter, the pack travels together while in summer part of the pack is denning and there is therefore a reduced hunting group. Large packs have been known to temporarily split into smaller groups to increase their hunting efficiency (Mech & Boitani 2003).

As wolves neither guard nor hide their kills, the optimum foraging strategy is therefore to hunt prey that the hunting group can consume to the fullest extent possible in a single feeding session (Jędrzejewski et al. 2002). The pack can then grow as long as sufficient food can be provided for all members (Mech & Boitani 2003).

Kill and consumption rates. Wolf kill rates vary depending on pack size and season. In the western Italian Alps, annual wolf kill rates were estimated to be 53 ± 16.5 red deer individuals/ 100 km^2, 83 ± 40 roe deer individuals/100 km^2 and 23 ± 7 chamois individuals/100 km^2 (Gazzola et al. 2007). Wolf consumption rates are difficult to determine as it depends on a variety of aspects such as pack size, season, prey availability etc. Wolf consumption rates were calculated in some European countries (Table 5.7).

Table 5.7. Wolf kill and consumption rates in Europe

Wolf kill rates	**Average pack size**	**Country**	**Source**
0.14 (per wolf/day)	5.5	Poland	Jędrzejewski et al. 2000
0.21 (per pack/day)	3.5	Poland	Jędrzejewski et al. 2002
0.67 (per wolf/day)	6 adults 4 pups	Finland	Gurarie et al. 2011
0.83 (per wolf/ day)	9-11 adults, 8 pups	Finland	Gurarie et al. 2011
Wolf consumption rate (kg/wolf/day)			
5.6	5	Scandinavia	Sand et al. 2008
4.98	9-11 adults, 8 pups	Finland	Gurarie et al. 2011
3.2	6 adults 4 pups	Finland	Gurarie et al. 2011

In some cases, prey can become locally very abundant (e.g. clustered in a restricted area) and consequently more vulnerable to predation. Wolves may then kill several animals over a short period of time, but consume little or none of the carcasses; this phenomenon is known as "surplus killing" (Mech & Peterson 2003). However, ‘surplus kills’ are not wasted; when not disturbed, wolves will cache parts of the carcasses and revisit the caches in times of low prey availability (Mech & Boitani 2003). Additionally, scavengers and opportunistic carnivores benefit greatly from such carrion when abandoned by wolves (Peterson & Ciucci 2003). The ability to make surplus kills and cache food sometimes trigger “massacres” among domestic animals, which are highly clustered and extremely vulnerable to predation because of their domestic nature and because they may be restricted in their movements, e.g. in a pen.

Predation studies in the countries of the Alpine Convention

We review here the results from studies carried out in countries that are party to the Alpine Convention, including information coming from regions outside the Alps. Within the Alps, the behaviour and feeding habit of wolves were mainly studied in the south-western Alps in Italy and France, where wolves have started to recolonise some 25 years ago.

France. The expansion of the Italian Apennine population and the crossing to the Southern Alps (Chapter 3.3.1) was facilitated by an increase of prey (red deer, wild boar) in the northern Apennine. Wolf predation in France has been relatively well studied since the recolonisation of the species in the country, due to the high degree of livestock depredation (Fluhr 2011). A number of packs have established territories in several protected areas of the French Alps. A few studies on their predation on wild and domestic prey have been carried out, especially in the Mercantour National Park (Parc de Mercantour). Summer was the limiting season for wolves in the Mercantour, when wild prey were more difficult to obtain as they moved into difficult and steep terrain. However, the abundant and relatively easily accessible domestic livestock allowed wolves to successfully continue their colonisation of the Alpine region (Espuno 2004).

Wolf diet studies in France were carried out since about 1995 (Fluhr 2011). Scats from nine packs (with sizes ranging from five to nine individuals) in the French Alps were collected between 1995 and 2009. These packs showed a relative uniformity in their predation with 76% of wild ungulates, 16% livestock and 8% of smaller prey, and the results of the analysis showed that variations in the diet of these packs were based on environmental factors such as the type and abundance of wild prey and in particular the type of livestock protection programmes applied in the region (Fluhr 2011). One of the packs showed a preference for domestic prey: livestock made up 43% of the pack's diet during the summer and 46% in winter in the Mercantour Protected Area. This high dependence on domestic animals is a consequence of livestock being present ten months per year in this PA (Fluhr 2011). Packs in other regions where livestock were present only in summer also predated on domestic animals but these made up only 12–29% of the overall diet (Fluhr 2011). Other packs such as those in the Haute Tinée and Vésubie-Tinée preyed on mouflon, chamois and ibex with greater frequency during winter (Fluhr 2011). In the Haute Tinée region, there was a net decrease in predation on mouflon between 1997 and 2007 possibly due to the decrease in mouflon population[9]. This led to a shift in diet and diversification of species predated on including wild boar, red deer and roe deer (Fluhr 2011).

Other studies in the French Alpine region also show that chamois, mouflon, roe deer, red deer, ibex, wild boar and domestic sheep are the wolves' main prey and are taken in varying proportions depending on seasonality and availability (Fig. 5.46, 5.47; Duchamp et al. 2012).

[9] Mouflon seem to be highly vulnerable to predation, probably as a consequence of their assumed history as domesticated animals. It was also observed in areas where lynx had access to mouflon population that the kill rate was so strong that it led in some cases to the extermination of (fenced) populations.

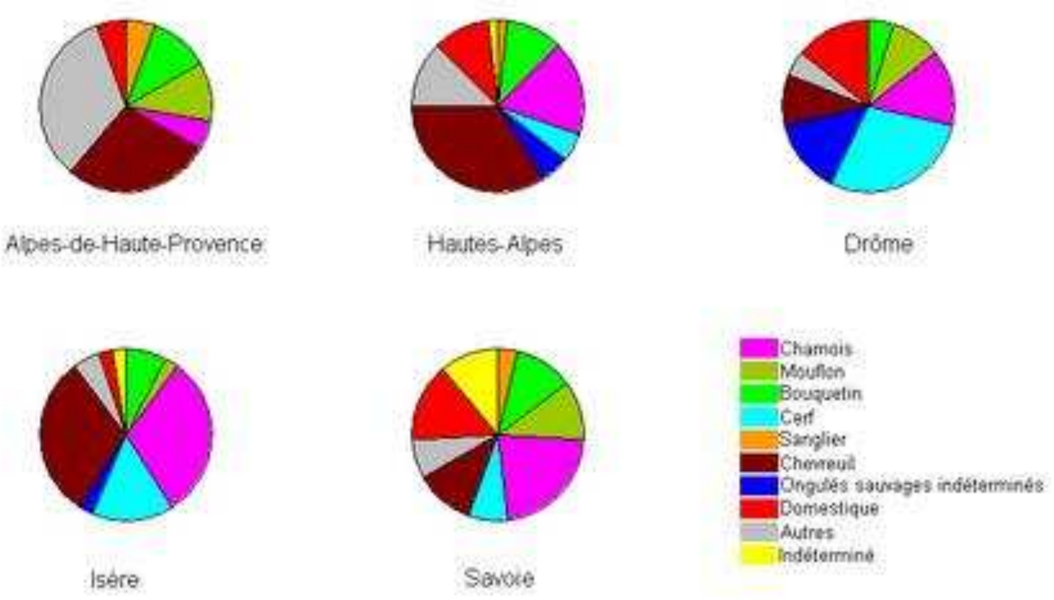


Fig. 5.46. Wolf diet in five French Alpine departments. Species names: chamois = chamois, mouflon = mouflon, bouquetin = ibex, cerf = red deer, sanglier = wild boar, chevreuil = roe deer, ongulés sauvages indéterminés = unidentified wild ungulates, domestique = domestic ungulates, autres = others, indéterminé = unidentified (Duchamp 2004).

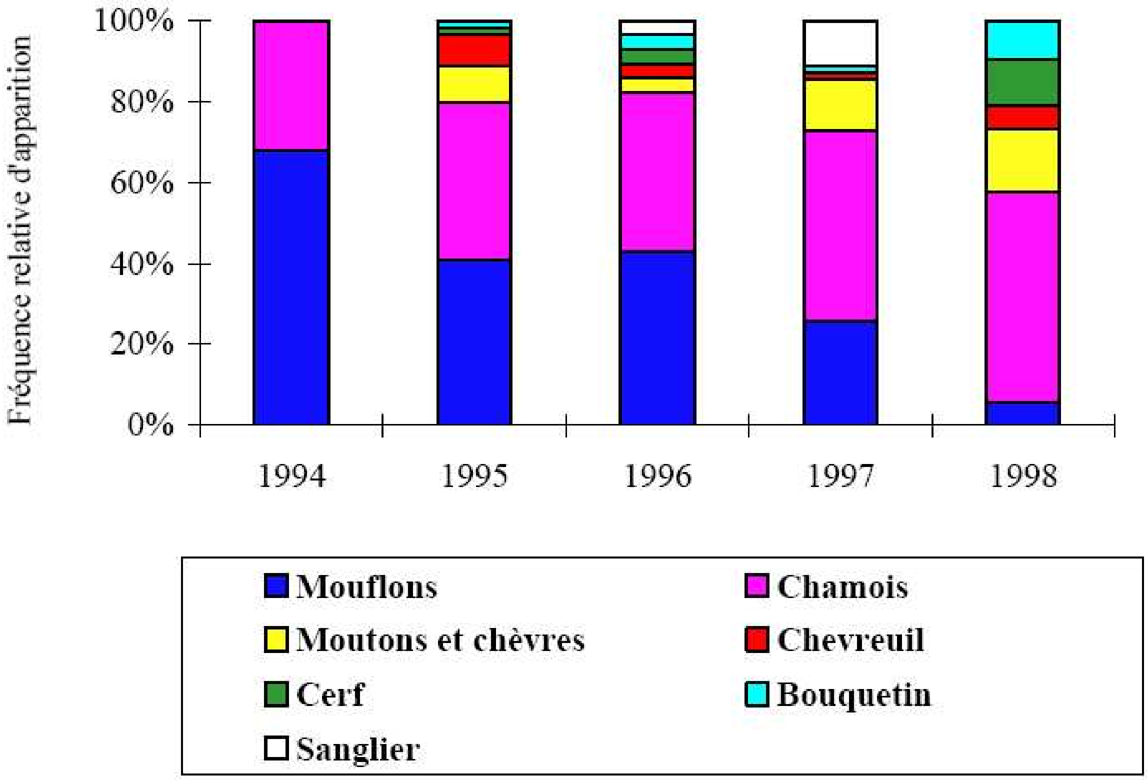


Fig. 5.47. Evolution of wolf diet in the Mercantour Protected Area between 1994 and 1998. Species: mouflons = mouflon, moutons et chèvres = sheep and goats, cerf= red deer, sanglier = wild boar, chamois = chamois, chevreuil = roe deer, bouquetin = ibex (Duchamp 2004).

Espuno (2004) found that chamois and mouflon were primary prey while roe deer, red deer and domestic sheep were secondary (seasonal) prey of some wolf packs in the Mercantour Protected Area. An analysis of 435 scats collected during the winters of 1999–2002 in the Western Alps, Valle Pesio (Italy/France) showed that wild ungulates were an important source of prey in the winter season with wild boar and red deer being highly selected for (Marucco et al. 2008).

While the wolves in France prey mainly on wild ungulates, in certain seasons and depending on wild prey availability, they can turn to domestic livestock. The seasonal importance of domestic livestock in the wolf diet has been shown in the Mercantour Protected Area (Fig. 5.48).

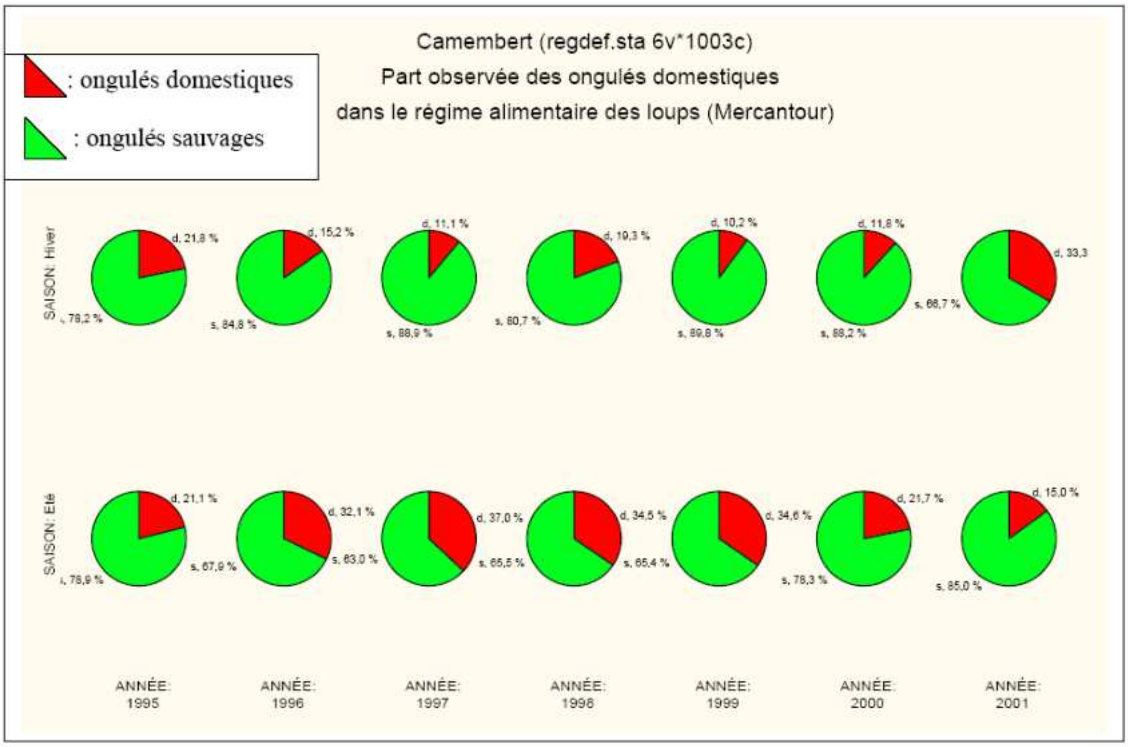


Fig. 5.48. Observed percentages of domestic ungulates in the wolves' diet in the Mercantour Protected Area and the evolution of the diet between 1995 and 2001. Green= wild ungulates, red= domestic livestock (Duchamp 2004). *Hiver* = winter, *Eté* = summer.

During the summer, in areas where transhumance was practised, wolves tended to rely on livestock. Due to increased and targeted protective measures, livestock depredation has been reduced in some areas. In some areas, wolves can engage in "surplus killing" and such behaviour was documented in the French Alps where wolves were recorded killing a greater number of domestic animals than they could consume (Lescureux 2002). The attacks on livestock in France consistently increased over the years (Fig. 5.49), from 36 sheep killed in 1993, 122 killed and 3 injured in 1994 and 359 were killed and 33 injured in 1995 (Anonymous 1996). There was a first peak in 2005 with 3,762 victims. For the next four years the average was 2,863 killed domestic livestock. Since 2009 numbers have steadily increased to 6,211 in 2013 and 8,226 in 2014 (Duriez et al. 2010, DREAL & DRAAF Rhône-Alpes 2011, ONCFS 2014a, DREAL 2015). Corresponding with the wolf distribution in France, the vast majority of these events occurred within the Alpine departements. The use of preventive measures such as guard dogs and electric fences are encouraged (Kaczensky et al. 2013d).

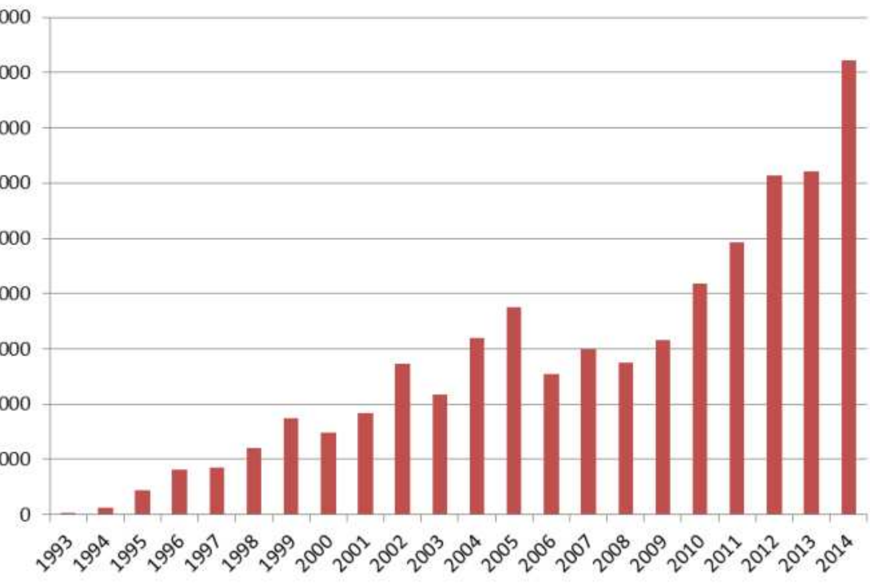


Fig. 5.49. Evolution of number of domestic livestock killed in France 1993-2013 (Duriez et al. 2010, DREAL & DRAAF Rhône-Alpes 2011, ONCFS 2014a, DREAL 2015).

Italy. As in the other Alpine countries, wolves in Italy generally prey on ungulates and their distribution and presence is predominantly in the western Italian Alps (Apollonio et al. 2010). An analysis of 848 wolf scats collected in the western part of Turin between December 1999 and November 2002, showed that wild ungulates were the most important prey, contributing 87.2% to the total, with cervids as the predominant species contributing 84.2% in winter and 54.3% in summer (Gazzola et al. 2005). Young and female red deer were selected for, contributing 29.4% and 58.8%, respectively, to the total amount of red deer consumed (Gazzola et al. 2005). Roe deer were found to be predated on more frequently than chamois and red deer. Results from a study by Ciampichini (2006) showed that in the winter of 2003–2004, 52.5% of the wolf diet consisted of roe deer, 21.2% of chamois, 16.4% of wild boar and 6.8% of livestock. In the winter of 2002–2003, 37.3% of the wolf's diet consisted of chamois, 30.1% of wild boar 19.5% of roe deer and 8.0% of livestock (Ciampichini 2006). During the summer of 2003, chamois made up 33.2% of the wolf's diet, along with 27.6% of livestock, 19.4% of roe deer and 8.6% of wild boar. In all seasons the rest of the diet consisted of other mammals such as red deer, mouflon, marmot, hare and fox (Ciampichini 2006). Based on two individuals in two different Alpine regions in Italy, Valle Pesio and Valle Susa, deer and wild boar were found to make up the majority of the wolf diet with their importance varying over the year (Marucco et al. 2010). This variation is likely to reduce the predation impact on ungulate populations (Marucco et al. 2010). In the winters of 2004–2005 and 2006–2007, wild boar were preyed on more frequently than deer while in the winter of 2005–2006, roe deer were more frequently preyed on (Marucco et al. 2010). During a study in the Piedmont Region, a total of 2,586 scat samples were collected between October 2004 and April 2008 (Regine 2008), 79% in winter and 21% in summer. The results of the study showed that in summer, 69.5% of the wolf diet consisted of wild ungulates while 31.9% consisted of livestock. In winter, around 70% of the diet is comprised of wild ungulates while livestock made up around 25% of the diet for the years 2004–2005 and 2006–2007 (Regine 2008). A similar study carried out in 2010–2011 showed that the diet of wolves in two study areas in the Piedmont Region was mainly composed of roe deer while chamois and wild boar were preyed on to a lesser extent (Rizzuto 2012). Palmegiani et al. (2013) found that the wolf's diet in summer comprised mainly of chamois while in winter chamois and roe deer were preyed on in relatively similar ratios (Fig. 5.50).

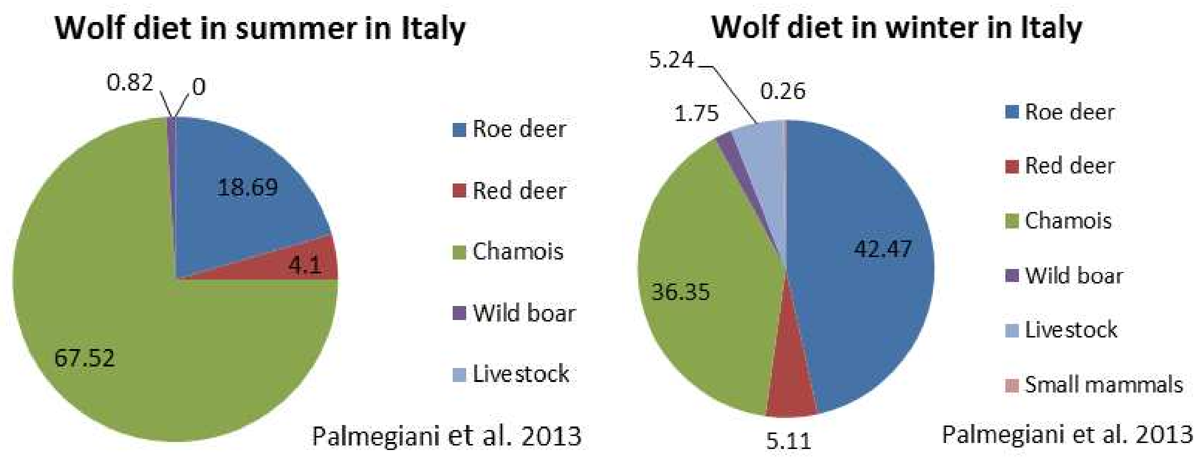


Fig. 5.50. Diet of wolves in Gran Paradiso National Parki, western Alps, Italy in the summer 2008 and winter of 2008-2009 based on scat analysis (Palmegiani et al. 2013).

Livestock depredation in the western parts of Turin, which were studied between December 1999 and November 2002, contributed 19.0% during the summer and 0.3% in winter to the wolves' diet. Overall, domestic prey accounted for 6.6% of the total wolf diet and largely due to the presence of unguarded livestock in the pastures during the months of May to October (Gazzola et al. 2005). In the Gran Paradiso National Park sheep and goat made 5.24% in winter 2008/2009 and were absent in the summer (Fig. 5.50; Palmegiani et al. 2013). In 2011, in the Piedmont region, 383 domestic animals, mostly sheep and goat, were preyed on by wolves leading to moderate compensation costs (Kaczensky et al. 2013d; Chapter 6.3.2). The number of attacks by dogs and wolves in the Piedmont region was also studied by Dalmasso et al. (2012). They found that sheep formed 79.4% of the domestic livestock victims (Fig. 5.51).

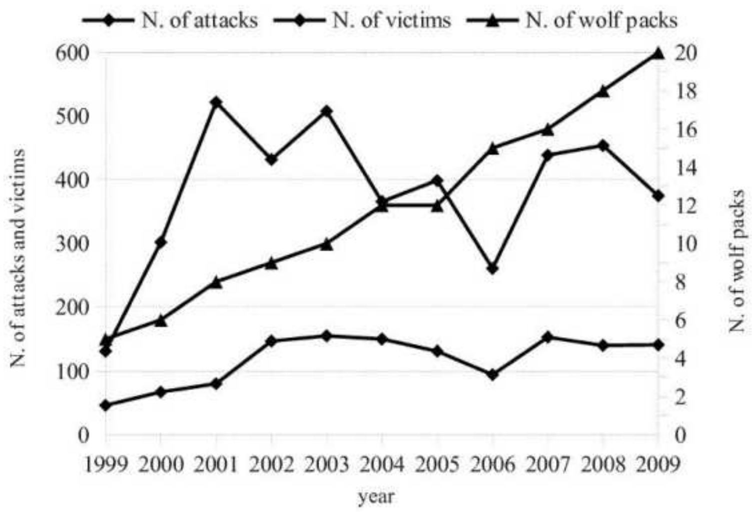


Fig. 5.51. The canids (wolf and dog) trend of damage on domestic animals and the trend of wolf packs in the Piedmont Region: number of attacks, number of victims (dead and injured) and number of wolf packs in the years 1999–2009. Sheep constituted 79.4% of victims, goats 16.8%, bovids 3.5%., equids 0.2% and shepherd dogs 0.1% (Dalmasso et al. 2012).

Switzerland. Compared to France and Italy, there is little information on wolf diet from Switzerland, as up to 2014, only one pack had established and all other resident or transient wolves were single individuals. Weber & Hofer (2010) studied the diet of wolves recolonizing Switzerland based on scat analysis. Following reports of livestock depredation, scat samples were collected between 1999 and 2006. The 81 scat samples originated from lone wolves as no packs were recorded in the country during the study period. Wild ungulates made up 65% of the wolves' diet. Red deer constituted the main prey with 32% (frequency of occurrence) and was taken according to its availability (Fig. 5.52). Roe deer were positively selected for and occurred in 21% of the scat samples. Chamois, ibex, wild boar and mouflon were each found to amount to less than 5% of the samples. The scats were collected in the scope of a depredation study, therefore the percentage of livestock in the diet is higher than what we would expect if we only considered randomly collected scat samples.

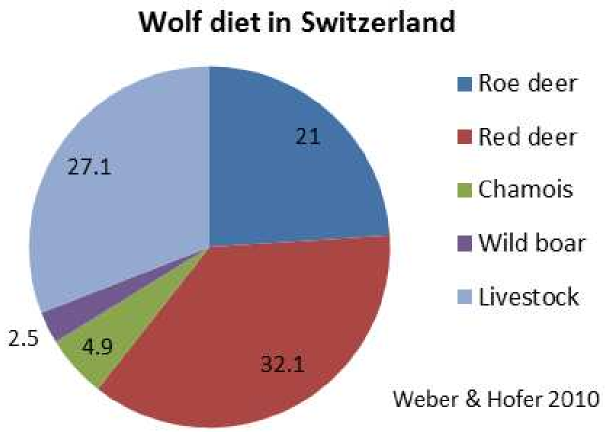


Fig. 5.52. Wolf diet in Switzerland based on frequency of occurrence in scats (Weber & Hofer 2010).

The number of livestock compensated as wolf kills has been recorded since 1994 when the first wolves immigrated into the Swiss Alps (Fig. 5.53). The number of individual wolves identified using genetic analysis increased continuously, with the first pack established in eastern Switzerland in 2012 (KORA, unpublished data). The recent decrease in livestock kills (Fig. 5.53) might be due to improved livestock protection measures.

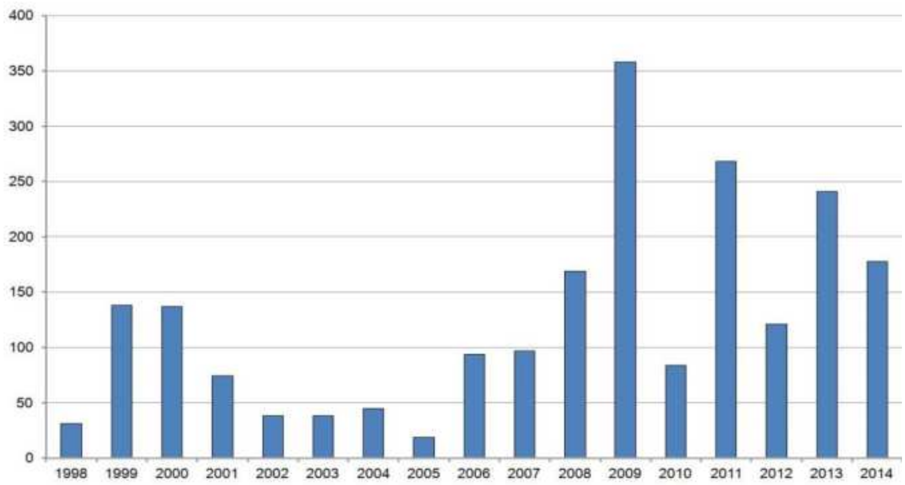


Fig. 5.53. Livestock (94% sheep, 5% goats, 1% others) compensated as wolf kills in Switzerland since the beginning of the recolonisation Source: KORA 2014.

Liechtenstein. No wolf or wolf kills were reported from Liechtenstein up to now.

Germany. So far wolves in the German Alps were single disperses which travelled into Bavaria. The main population is in the north-eastern part of the country which is outside of our area of concern.

The authors of a study in the Saxony region of Germany, based on an analysis of 192 scats collected between 2001 and 2003, found that almost all samples contained remains of wild ungulates. The importance of this group of prey species is quantified as representing 96% of the biomass consumed with 56% being from red deer, 21% from roe deer and 16% from wild boar (Ansorge et al. 2006). Other studies found different percentages of prey species in wolf scats (Fig. 5.54).

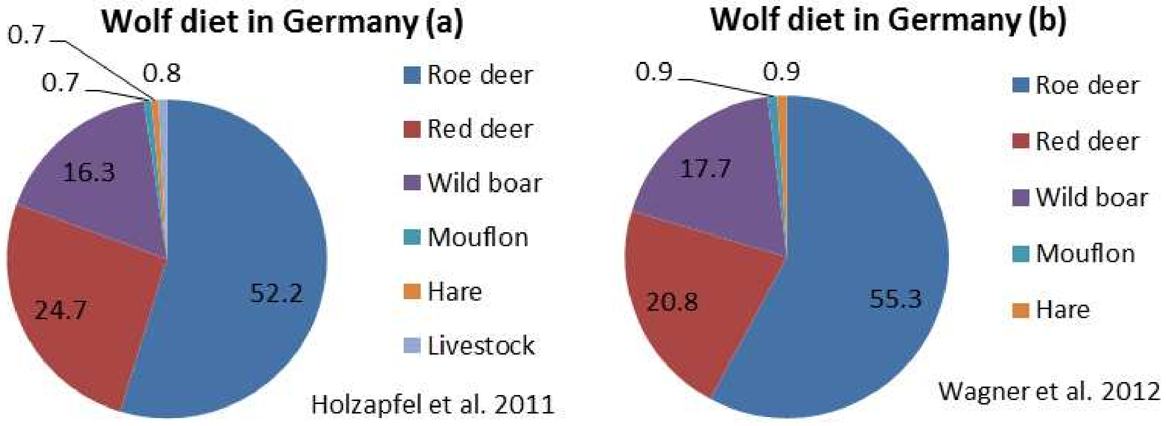


Fig. 5.54. Wolf diet based on two studies in Germany based on percentage of biomass consumed (Holzapfel et al. 2011, Wagner et al. 2012).

In general, livestock depredation is relatively low compared to other European countries (Kaczensky et al. 2013d; Chapter 6.3.2). In 2011, wolves killed 225 domestic animals in all of Germany (Kaczensky et al. 2013d).

In the case of Germany, it is important to look at the feeding ecology of wolves in Poland as the wolves are thought to have crossed over from Poland. The ungulate composition and therefore the wolf diet appear to be similar. The wolves' primary prey species in the Tatra National Park (Slovakia and Poland) were red deer and roe deer, and wild boar were preyed on to a lesser extent (Findo & Chovancova 2004). While wolves tended to prey mostly on wild ungulates, livestock depredation occurred during the study period between 1998 and 2004 with 591 confirmed incidents (Gula 2008). These attacks mostly occurred during the grazing season between the months of May and October to November. However, wolves tended to choose wild ungulates (83.2% frequency of occurrence in 719 scat samples) despite the availability of easily obtainable domestic prey (7.8%). Dogs were preyed on more often than sheep (Gula 2008). An analysis of 144 wolf scats in the Białowieża National Park in Poland showed that cervids (mostly red deer) were the principle prey taken (91% biomass) during the months of October to April and this percentage decreased during the months of May to September to 77% with an increase of wild boar intake (Jędrzejewski et al. 1992). Further studies in the same areas showed that wolves annually killed 72 red deer, 16 roe deer and 31 wild boars over 100 km^2 (Jędrzejewski et al. 2002).

Livestock depredation cases by wolves in the Polish Carpathian Mountains were studied between 1998 and 2004 and found to be opportunistic in function of wild prey availability and domestic animal abundance (Gula 2008). Inadequate husbandry practises were also found to be an important factor in the rate of livestock depredation (Chapter 6.3.2), although the tradition of protecting herds has never been lost in the Carpathian Mountains as much it was forgotten in the Alps.

26 sheep were kills by a transient wolf in 2010 in the Bavarian Alps, and 3.670 € of compensation paid. Emergency preventive measures (night pens and change of pastures) were implemented at the site of the attacks. Further preventive measures (especially changing the common husbandry system) are recommended, but not implemented yet (M. Wölfl, pers. comm.).

Austria. Although the wolf's historical range extends into the Austrian Alps, the number of wolves present in this region is low. Information on predation is only anecdotal and concerns mostly live-stock depredation. There have been some livestock depredation cases in Austria attributed to wolves (Kaczensky & Rauer 2013). In 2009, about 70 sheep were killed or wounded or went missing. In 2010, 18 sheep and goat and two calves were killed or wounded while about 90 sheep were missing at the end of the season (Kaczensky & Rauer 2013). In 2011, 14 sheep and goats and one calf were killed or wounded (Kaczensky & Rauer 2013). A pilot project to promote the use of protective measures such as electric fences and guard dogs was launched in 2012 (Kaczensky & Rauer 2013; Chapter 6.3.2).

Slovenia. Information on wolf diet and predation concerned mainly the southern part of Slovenia, outside the Alps. The main prey base of wolves in Slovenia includes red and roe deer, wild boar and chamois (SloWolf 2014). The improvement of management of wild ungulate species was addressed over the course of six workshops within the SloWolf project. The result consisted of management recommendations, which are implemented since 2013 in the yearly game management plans for districts (M Jonozovič, pers. comm.).

The data available show that, like in some other Alpine countries, the key prey species of the wolf in Slovenia are red deer and wild boar. One study using a sample size of 30 wolf scats, reported that wolves predated primarily on wild cervids (85% of consumed biomass) and young wild boar (5% of consumed biomass) (Krofel & Kos 2010).

Although the wolves' population is relatively low and wild ungulates such as red and roe deer are abundant, wolves in Slovenia appear to prey on domestic livestock increasingly (Fig. 5.55; van Liere et al. 2013). Domestic animals constitute about 10% of the wolf's consumed biomass (Krofel & Kos 2010, van Liere et al. 2013). A total of 107 depredation cases were recorded in the Alps since 2006 (M. Jonozovič, pers. comm.). Although livestock protection is encouraged, most flocks have little to no protection (Kaczensky et al. 2013d). Between 1995 and 2003, livestock depredation cases were studied by Adamic et al. (2004). About 22.1% of all cases of depredation were attributed to wolves while the rest were caused by bear and lynx (Adamic et al. 2004). However, most cases occurred outside the Alps (Fig. 5.56; Černe et al. 2010).

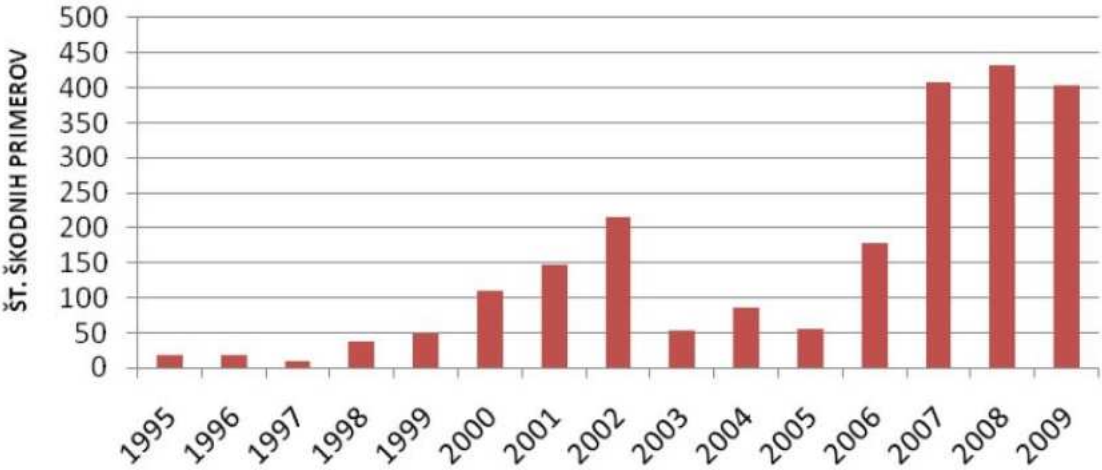


Fig. 5.55. Development of number of attacks per year in Slovenia (Černe et al. 2010). About 4 animals were killed per attack, equalling 1500–1800 for the most recent years shown (Černe et al. 2010).

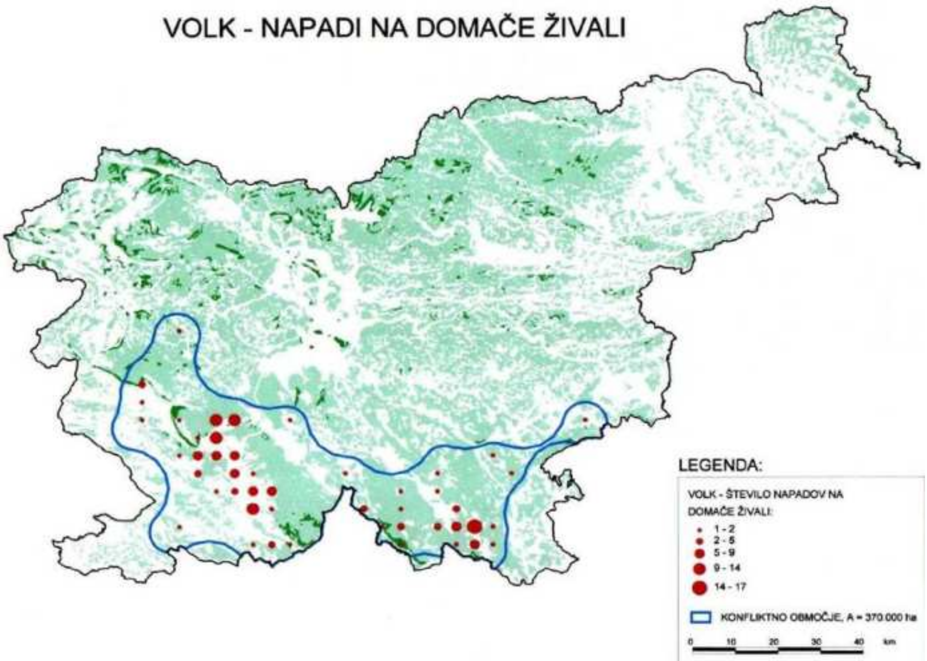


Fig. 5.56. Distribution of depredation cases 1995–2001 in Slovenia (Černe et al. 2010). Most cases were reported from outside the Alps.

Impact of wolves on prey populations

The number of prey required on a yearly basis by wolves is dependent on the pack size and season. As most studies are carried out in winter, it is difficult to have actual numbers of individual prey killed per year. Only few studies are carried out during the non-winter seasons. Mech (1971) estimated annual kill rates at 15–19 adult deer per wolf per year assuming that 20% of their annual diet consists of other prey. In Sweden, a pack killed 66 moose in four months in summer; however the pack size varied between 2 and 9 wolves (Sand et al. 2008). Gazzola et al. (2007) estimated the annual consumption of red deer by wolves in the western Italian Alps at 53 ± 16.5 individuals/ 100 km^2, roe deer at 83 ± 40 individuals/100 km^2 and chamois at 23 ± 7 individuals/100 km^2.

Wolves do not appear to have clear prey preference, consuming available prey as well as carrion and garbage (Mech & Peterson 2003). However, prey specialisation can occur in some individual wolves locally, especially in areas where one prey species is particularly abundant. In the Gran Paradiso National Park in Italy, roe deer consumption was higher in winter than in summer while predation on chamois showed an inverse trend (Palmegiani et al. 2013).

Wolves tend to prey on animals which have a lower physical condition and would be naturally selected out. A number of characteristics that can determine the vulnerability of prey to wolf predation were listed by Mech & Peterson (2003) based on the findings of numerous studies and include: time of year (during the rut, males are often preyed on due to decreased vigilance), age (calves and old animals selected for), weight (light animals often taken), disease, parasites, injuries and abnormalities, genetic condition and parental age (offspring of older parents preyed on less frequently).

A number of primary positive effects of wolf predation on their prey and ecosystem include the culling of unfit animals, control or limitation of prey numbers, stimulation of prey productivity and increase in food for scavengers (Mech & Peterson 2003, Peterson & Ciucci 2003). As wolves appear to take out individuals in lower physical condition, their effect on the prey population is reduced as they seem to cause mainly compensatory mortality. Such selective culling could even have a positive effect on the average health of the prey population. In general, wolves have a high plasticity and use prey and resources at their availability. Unlike lynx (which eat only animals they hunt themselves; Chapter 5.5.1), they are less constrained by territories and can travel longer distances in search of food during periods of low prey abundance. They are also capable of surviving on small species until other larger prey become available (Mech & Peterson 2003).

Experience with the impact of wolves on their (wild) prey populations in the Alps is still limited and so far restricted to the western Alps, where wild ungulate densities are lower and the populations are less managed than in the eastern Alps (Chapters 5.3 and 6). Indirect and longer-term effects may include changes in the structure of prey herds (age, sex, condition) but also changes in behaviour, movement, and local distribution. Wolf predation may also influence the effect of herbivores on vegetation structure.

5.6. Discussion and conclusions

While the decrease in wild ungulates, the destruction of forests and the increase in livestock numbers (negatively affecting the forests, out-competing the wild ungulates and leading to more human-carnivore conflicts) played – besides the direct persecution – a huge role in the disappearance of large carnivores from the Alps (Chapter 3), the reversion of the these processes now aids their return beyond the legal protection. Forest areas have grown again, wild ungulates have reached what is possibly their highest abundance in the Alps since time immemorial and sheep numbers in France, Italy and Germany (data for whole countries) are nowadays lower than in 1960. The human population has almost doubled since 1850, but became more concentrated in peripheral and inner-Alpine centres. The current level of habitat fragmentation in the Alps seems to have limited negative effects on wolf, lynx and wild ungulates. On the one hand, the Alps still have a rather low level of fragmentation within Europe (Jaeger et al. 2011), on the other hand, both large carnivores and the wild ungulates also occur in areas outside the Alps which are more heavily fragmented.

The high abundance in wild ungulates reduces the conflicts with livestock holders in comparison with the late 19th century, as both, wolf and lynx, seem to prefer wild ungulates over livestock, if wild prey is sufficiently available. At the same time, sheep farming in the western parts of the Alps is decreasing steadily as this “business” is only viable if it can be done at low costs and if it is supported through subsidies. Indeed, the sheep husbandry is heavily subsidised in all Alpine countries, but strongest in Switzerland (Giannuzzi Savelli et al. 1997). The financial aid and subsidies in Germany, France, Austria and Switzerland for mountain agriculture with regard to the return of the large carnivores was recently reviewed by Meschnig (2014), who has also compiled the total amount of aid provided in these four countries (Table 5.7).

Table 5.7. Financial aid (subsidies) in four countries of the Alpine Convention for livestock husbandry (all species) divided into “Alpengrünland” (agricultural areas within the Alps) and “Almfutterfläche” (Alpine grazing areas). Source: Meschnig (2014) based on data from Ringler (2009).

	Agricultural areas in the Alps			Alpine grazing areas		
Countries	ha	€/ha	Total amount [mill. €]	ha	€/ha	Total amount [mill. €]
Germany	45.000	622	28	36.000	666	24
France	700.000	371	260	651.000	369	240
Austria	660.000	363	240	505.000	243	123
Switzerland	570.000	342	195	537.000	287	154
Sum	1.984.000	364	723	1.729.000	313	541

The motivation for supporting sheep husbandry is threefold: (1) support for the agricultural sector and marketing for agricultural products in general, (2) support for economically disadvantaged regions (with the Alps considered as such in all countries of the Alpine Convention), and (3) support for extensive approaches in agriculture and measures to preserve the countryside (prevent reforestation), through conservation-oriented payments. Still, many of the sheep owners cannot live from sheep farming alone; the majority of owners today are not professional sheep farmers. As a consequence, the financial loss from livestock depredation and the requirements of financial and labour investments into livestock protection measures may be perceived as even more painful. Thus, conflicts with livestock holders are still prominent in spite of the totally changed ecological and economic conditions compared to the 19th century, and greatly influence regional and national political decisions. Furthermore, the impact of large carnivores on the populations of wild ungulates also leads to reinforced conflicts with hunters. Thus, while conditions in the landscape are again favourable for the presence of large carnivores, conflicts with human activities and interests are still the crucial element.

6. Wildlife management

6.1. Organisation of large carnivore management in the Alpine countries

6.1.1. Legislation

International treaties

At an international level, lynx and wolf trade are regulated by the Convention on the International Trade in Endangered Species CITES, where European sub-species are listed in Appendix II (CITES 2014). This appendix includes species which are not threatened with extinction as such, but are prone to extinction if wildlife trade is unregulated.

Species such as lynx and wolf have populations distributed across several countries and can have large individual home ranges, often >100 km^2 (Linnell et al. 2008; Appendix III of this report). Therefore, (legal) instruments to protect these species need to be coordinated at an international level, and several international treaties have been established to address transboundary conservation (Box 6.1). All the countries in the Alpine arc are signatories to the Bern Convention and the recommendations are implemented in their management plans (Trouwborst 2010). All large carnivores in Europe are covered under Annex II and IV of the Habitats Directive except (in the Alps) for the non-EU countries Switzerland and Liechtenstein (Linnell et al. 2008). Article 12 of the Habitats Directive requires member states to not only adopt a comprehensive legal framework for the concerned species, but also to implement concrete measures to ensure effective conservation actions (Trouwborst 2010). The Alpine countries are also signatories to the Bonn Convention which addresses the threats faced by migratory species. Legal instruments such as the Bern Convention can restrict the freedom of a country to manage listed species may lead – as is the case in Switzerland with regard to wolf management – to severe controversial political discussions.

Under European law, the wolf is listed as a priority species in Annex II which requires the designation of protected areas which are part of the Natura 2000 network (Box 6.1; Table 6.1; Trouwborst 2010).

Table 6.1. Legal status of lynx and wolf in Europe (Trouwborst 2010).

Legal Instrument	Lynx	Wolf
Convention on the Conservation of European Wildlife and Natural Habitats (Bern Convention)	Appendix III	Appendix II
EU Habitats Directive	Annex II Annex IV	Annex II (priority) Annex IV

National legislation lynx

Lynx is granted protection in all Alpine countries, however there are exceptions mainly concerning livestock raiding individuals which are removed in France and Switzerland. In most of the countries, national authorities are in charge of lynx conservation and management. However, in Germany, Austria and partly also in Switzerland, power is delegated to the regional authorities (Table 6.2).

Box 6.1. Legal instruments of interest to lynx and wolf conservation in the Alpine arc in order of importance

Bern Convention	The Convention on the Conservation of European Wildlife and Natural Habitats, also known as Bern Convention, was adopted in September 1979 in Bern, Switzerland, and came into force in June 1982. As per today, it has been ratified by 50 members, as reported on the official website of the Council of Europe (www.coe.int/bernconvention). Its overall aim is the protection of natural habitats and species, with emphasis on endangered and vulnerable species. It contains four appendices where species are classified as per their degree of vulnerability.
CITES	CITES, the Convention on International Trade in Endangered Species of Wild Fauna and Flora, is an international agreement between governments to ensure that international trade in specimens of wild animals and plants does not threaten their survival. It came into force in 1975. (www.cites.org)
EU Habitats Directive	The EU Habitats Directive also known as the 'Council Directive 92/43/EEC on the Conservation of natural habitats and of wild fauna and flora' is a European Union directive adopted in 1992. It has four annexes with the aim of protecting habitats, species requiring designation of "Special Areas of Conservation", species in need of strict protection and species for which wild harvest is restricted by European Law. (ec.europa.eu/environment/nature/legistation/habitatsdirective/index_en.htm)
Natura 2000	The Natura 2000 network was established under the Habitats Directive and comprises of a series of protected areas within the European Union (Emerald-Network for Switzerland and Liechtenstein).
Bonn Convention	The Bonn Convention - originally named the Convention on the Conservation of Migratory Species of Wild Animals (CMS) - came into force in 1983. It has been ratified by 120 parties since its conception as per the official website (http://www.cms.int/en/parties-range-states). This international treaty aims to conserve terrestrial, avian and marine species which are known to undertake migratory journey as part of their natural history.

Table 6.2. Legal status of lynx, restrictions to the status and authority in charge for lynx conservation and management in the Alpine countries. Sources: FR: Marboutin 2013b; IT: Anonymous 2012; CH: BUWAL 2004a, von Arx & Zimmermann 2013; FL: Fasel 2001; DE: Kaczensky et al. 2013b; AT: Huber et al. 2001, Kaczensky et al. 2013c; SL: M. Jonozovič, pers. comm., Koren et al. 2006, Stanisa et al. 2001.

Country	Legal status	Management interventions	Authority in charge
France	Strictly protected.	Selective removal of stock raiding individuals.	Ministère de l'écologie, du développement durable et de l'énergie.
Italy	Strictly protected.		Ministry of Environment; enforcement of general policies by the local administrations.
Switzerland	Strictly protected.	Selective removal of stock raiding individuals. Criteria for population regulation under discussion.	Federal Office for the Environment FOEN; the cantons for the implementation of the lynx concept (Chapter 6.1.2).
Liechtenstein	Strictly protected.		Amt für Umwelt (former Amt für Natur, Wald und Landschaft).
Germany	Subject to the hunting law, but year-round closed season.		Nature conservation authorities of the federal states, but the respective hunting authorities also have a responsibility.
Austria	Mainly subject to the district's hunting laws, but year-round closed season.	Special permits to shoot a lynx to be issued by the district authorities on request, so far never used.	Hunting and nature conservation authorities of the provinces.
Slovenia	Strictly protected since 2004 (before quota hunting from October to February)		Ministry of Environment and Spatial Planning.

National legislation wolf

Wolf is also strictly protected in all Alpine countries. This status is however subject to restrictions in order to reduce conflicts with livestock breeding. In Switzerland, livestock raiding individuals are selectively removed. In France and Slovenia, exceptional culls are permitted (Table 6.3).

Table 6.3. Legal status of wolf, restrictions to the status and authority in charge for wolf conservation and management in the Alpine countries. Sources: FR: Marboutin 2013a; IT: Boitani & Marucco 2013; CH: BAFU 2010, von Arx & Manz 2013; FL: N. Nigsch, pers. comm.; DE: Reinhardt 2013; AT: Kaczensky & Rauer 2013; SL: M. Jonozovič, pers. comm., Majić Skrbinšek 2013.

Country	Legal status	Management interventions	Authority in charge
France	Strictly protected.	Removal of stock raiding individuals (*tir de défense*). A yearly defined number of individuals are removed (*tir de prélèvement*).	Ministère de l'écologie, du développement durable et de l'énergie.
Italy	Strictly protected.	No derogation has ever been requested for culling under article 16 of the Habitat Directive.	Ministry of Environment, however regions in charge of management activities (Chapter 6.1.2).
Switzerland	Strictly protected.	Selective removal of stock raiding individuals. Criteria for population regulation if predation impact is too high are currently discussed.	FOEN; the cantons for the implementation of the wolf concept (Chapter 6.1.2).
Liechtenstein	Strictly protected.		Amt für Umwelt.
Germany	Strictly protected under the jurisdiction of the Federal Nature Conservation Act.[10]		Nature conservation authorities in the Länder. In some Länder the regional ministries of the environment are in charge, in other Länder responsibility is further delegated to the district administrations.
Austria	Wolf is mainly subject to the district's hunting laws, but year-round closed season.		Hunting and nature conservation authorities of the provinces.
Slovenia	Strictly protected since 2004 (before quota hunting from October to February).	Exceptional culls permitted to decrease conflicts with agriculture.	Ministry of Environment and Spatial Planning.

6.1.2. Management plans

European (population) level management plans

The European Commission started to introduce the population approach to large carnivore management, within the scope of the Habitats Directive. In 2005, it awarded a tender for the development of "Guidelines for population level management plans for large carnivores in Europe". The Guidelines, published in 2008 (Linnell et al. 2008), provide recommendations on how the concept of the Favour-

[10] In Saxony the wolf became subject to the hunting law with a year-round closed season in September 2012, resulting in a shared responsibility between nature conservation and hunting authorities (Reinhardt 2013).

able Conservation Status FCS of the Habitats Directive can become operational for large carnivores (Appendix IV). As for these species, FCS is more likely to be achieved on the level of the populations rather than the countries, the Guidelines stimulate transboundary management of large carnivores and advice on the development and content of population management plans (Box 6.2).

Box 6.2. Topics that a transboundary management plan should contain from the "**Guidelines for population level management plans for large carnivores in Europe**" (Linnell et al. 2008). Three sections are proposed, focusing on background information, a formulation of measurable, time specific and spatially explicit objectives and targets, and a set of actions that are needed to achieve these objectives. Further explanatory notes to each of the proposed chapters and subchapters are given in the Guidelines (Linnell et al. 2008).

1. Background
    - 1.1. Population definition
    - 1.2. Management units
    - 1.3. Population description
    - 1.4. Habitat description
    - 1.5. Continental context
    - 1.6. Current management
        - 1.6.1. Legal status and management regime
        - 1.6.2. Damage and conflicts
        - 1.6.3. Obstacles to conservation
        - 1.6.4. Conservation status
2. Definition of goals and objectives
    - 2.1. Statement of overall vision
    - 2.2. Measurable objectives
        - 2.2.1. Favourable reference population
        - 2.2.2. Favourable reference range
        - 2.2.3. Population goals
        - 2.2.4. Success criteria
        - 2.2.5. Connectivity and expansion
        - 2.2.6. Spatial aspects of management
3. Actions
    - 3.1. Maintaining range and population size
    - 3.2. Maintaining and enhancing connectivity
    - 3.3. Adapting legislation
    - 3.4. Ensuring adequate wild prey base, natural food supply and habitat quality
    - 3.5. Damage control and conflict resolution
    - 3.6. Coordinating harvest/control of carnivores
    - 3.7. Enforcement
    - 3.8. Cross-border exchange of experience among stakeholders and interest groups
    - 3.9. Institutional coordination of management authorities
    - 3.10. Coordination of monitoring and scientific research programs
    - 3.11. Ensuring sectorial coordination within and between the countries
    - 3.12. Monitoring efficacy of implemented management measures

The Guidelines were adapted according to the input from country workshops and discussions held in the Habitats Committee of the European Commission and the Committee's Scientific Working Group. The final version was presented at the "Pan European Conference on Population Level Management Plans of European Large Carnivores" from June 10-11, 2008 in Postojna, Slovenia. Patrick Murphy, Head of the Directorate-General (DG) Environment of the European Commission at the time, stated: "*These guidelines represent best practice for the management of large carnivore populations and DG*

Environment accordingly recommends them to the authorities in the Member States. The guidelines are not legally binding but do constitute a reference point against which DG Environment will monitor actions taken by the Member States in fulfilment of their obligations under the Habitats Directive".[11]

In 2012, the DG Environment of the European Commission launched an initiative for the conservation and sustainable management of large carnivores based on dialogue with, and involvement of, relevant stakeholders[12]. As a supporting document and basis for improving the implementation of the EU large carnivore policy under the Habitats Directive, key management actions for large carnivore populations in Europe were developed (Boitani et al. 2015). The document contains three levels of actions: (1) A set of 11 cross-cutting actions that are of general importance for large carnivore conservation in Europe across species (wolf, lynx, brown bear and wolverine) and populations; (2) A set of 6–8 key priority actions for each species that are applicable to the majority of its European populations, and (3) 2–3 specific priority actions for each population (Box 6.3).

The actions are meant as a guidance and voluntary agenda for national authorities responsible for implementing the Habitats Directive and for stakeholders who might take an interest in, and have the resources to implement some of the proposed measures (Boitani et al. 2015). Interest groups and EU Member States had the chance to comment on draft versions of the document. Their inputs were taken into account in the final version.

Management plans for the entire Alps

Lynx. In 2003, a Pan-Alpine Conservation Strategy PACS for the lynx was published (Molinari-Jobin et al. 2003) under the hospice of the Bern Convention. The strategy was elaborated by the SCALP expert group (Chapter 4.1) and proposed standards aimed at boosting transboundary activities and co-operation from local to international levels. The goal of the strategy was "*to establish and maintain, in co-existence with people, a vital lynx population covering the whole of the Alpine arc*" (Molinari-Jobin et al. 2003). This goal was then specified in four objectives:

1. The lynx populations in Slovenia and Switzerland maintain their vitality and must be helped to expand;
2. The populations in Slovenia and Switzerland are joined through colonisation of the area in between (Alps of Austria, Germany, Italy and Liechtenstein);
3. This unified population in the central Alps is allowed to expand to the north-east (Austria) and the south-west (France, Italy);
4. Gene flow is assured between the Alpine sub-populations and the population of Slovenia and Croatia, the population of the Jura Mountains and the population of the Bohemian/Bavarian forest.

To operationalise these objectives, actions on the pan-Alpine level as well as for each country were proposed (Molinari-Jobin et al. 2003).

[11] http://ec.europa.eu/environment/nature/conservation/species/carnivores/pdf/guidelines_for_population_level_management_ec_note.pdf
[12] http://ec.europa.eu/environment/nature/conservation/species/carnivores/index_en.htm

Box 6.3. Titles of the cross-cutting actions for large carnivores in Europe, key priority actions for lynx and wolf in Europe, and priority actions for the Alpine lynx and Alpine wolf populations. The actions are further described in Boitani et al. (2015).

Cross cutting actions – across species and populations	• Preventing fragmentation of habitat and reducing disturbance associated with infrastructure development • Reducing large carnivore depredation on livestock • Integrating large carnivore management needs into wildlife and forest management structures • Evaluating social and economic impacts of large carnivores • Improved transboundary coordination of large carnivore management • Standardisation of monitoring procedures • Managing free ranging and feral dogs to reduce hybridisation with wolves and other conflicts • Law enforcement with respect to illegal killing of large carnivores • Genetic reinforcement of small populations of lynx and bears • Institutional capacity building in wildlife management agencies • Developing best practice for large carnivore based ecotourism
Actions for all lynx populations	• Population-level and national management plans • Intra- and inter-population connectivity and fragmentation • Standardised, robust quantitative monitoring of lynx populations • Health monitoring and genetic reinforcement of small, inbred populations • Habitat conservation and environmental impact assessments • Integrate lynx predation impact into wildlife management practise
Specific actions for the Alpine lynx population	• Pan-Alpine and integrated conservation and management of lynx • Genetic reinforcement • Assisted merging of subpopulations
Actions for all wolf populations	• Standardised census and monitoring of wolf population • Transboundary cooperation and population-level Management Plan • Prevention and compensation measures to reduce livestock depredation. • Measures against illegal killing and control of poison baits • Control of free-ranging dogs and wolf-dog hybridization • Habitat fragmentation and connectivity • Education, information and data accessibility
Specific actions for the Alpine wolf population	• International Alpine Wolf Committee • Spatial models for managing the wolf population above the Favourable Conservation Status (FCS) • Quality improvement and correct use of livestock guarding dogs (LGD)

Wolf. In 2006, the Ministries of Environment of Italy, France and Switzerland signed an "italo-franco-suisse collaboration protocol for the management of wolf in the Alps" (Ministerio dell'Ambiente e della Tutela del Territorio et al. 2006). Taking into consideration the framework set by Habitats Directive and Bern Convention (Chapter 6.1) as well as the existing national management plans (see below), the Environmental Ministers declared their common goal, to "*re-establish and preserve viable wolf populations in the Alps in coexistence with people and notably mountain farming, consider the ongoing natural recolonization, concentrate their conservation measures on the population level and therefore reinforce the transboundary cooperation between the three countries, and be aware of preserving the adequate development of alpine zones as a result of livestock breeding*". The Ministers furthermore decided to organise periodical official meetings on different levels (national authorities and technical group, respectively) between the countries concerned in order to coordinate policies related to wolf management on one hand and to exchange information and experience on the other hand (Ministerio dell'Ambiente e della Tutela del Territorio et al. 2006).

National management plans for lynx

Switzerland is the only Alpine country that has elaborated and implemented a lynx management plan. The first "Swiss Lynx Concept" was endorsed in August 2000 and was revised in 2004 (BUWAL 2004a). It defines the general conservation and management goals, the co-operation between the Federal Office for the Environment FOEN and the cantons, and criteria for interventions. To harmonise the implementation of the concept, the country was divided into 8 management compartments. For each compartment, an inter-cantonal commission coordinates the decision-making with regard to monitoring, prevention and intervention measures. In June 2012, the revised Swiss hunting ordinance was enacted and currently, the concept is revised as well. It will define criteria for lynx regulation if the predation impact is too high. The removal of "problem" lynx, individuals killing too much livestock, is already possible under the current management plan (see Box 6.4).

Box 6.4. Swiss lynx concept: criteria for removing individual "problem" individuals (BUWAL 2004a).

- At least 15 domestic animals must have been killed within a 5 km radius
- This number can be reduced to 12 in the following cases: if animals have already been killed and eaten in the previous year; the lynx was not killed within the allocated shooting period; the killings continued even after the lynx was shot
- A total of 60 days is allocated to shoot the problem individual
- This duration can be extended by 30 days if new livestock damage is recorded

In Germany, Austria and Slovenia there have been intentions to develop a national lynx management plan, however, they do not exist yet:

Germany has a framework document by the Federal Agency for Nature Conservation (BfN) on how to deal with management issues concerning lynx, wolf and bear (BfN 2010) some parts of which have been published elsewhere (Kaczensky et al. 2009, Reinhardt et al. 2012).

Austria. A coordination board for bear, wolf and lynx management in Austria (KOST[13]) consisting of representatives of the hunting and natural conservation authorities of the provinces, the bear advocates and selected external experts meets twice a year to review and discuss management issues regarding large carnivores in Austria. A management plan for lynx in Austria however does not exist (Kaczensky et al. 2013c).

Slovenia. A national lynx management plan for Slovenia is in development. The document is expected to formulate and formalise the management goals and facilitate the implementation of a planned population augmentation project (Kos & Potočnik 2013). It is currently in the final stage and expected to be adopted by the Slovenian Government in 2015 (M. Jonozovič, pers. comm.).

[13] http://www.vetmeduni.ac.at/de/fiwi/dienstleistungen/koordinierungsstelle-fuer-den-braunbaeren-luchs-und-wolf/

National management plans for wolf

The development of a national wolf management plan was addressed in all Alpine countries, however the actual realisation and particularly the implementation of such plans differ. In federal states like Germany, Austria, Italy, and Switzerland, the nation-wide management of wolf (as well as of the other large carnivore species) is challenging because many responsibilities with regard to wildlife management are delegated to the regions. Although the legal status of the large carnivores implies responsibilities at national level, the implementation can be difficult because in federal states, the central authorities do not directly control the implementation agencies and institutions. The state of art in each country is briefly described here:

France. As early as 1993, an initial action plan was endorsed by the Ministry of Environment in the Mercantour National Park. This plan was revised and extended in the frame of two LIFE projects (Ministry of Agriculture and Fisheries & Ministry for Regional Planning and the Environment 2000).

In the "Wolf Action Plan 2004–2008" (MEDD & MAAPAR 2004), the French government defined an approach aiming to balance the needs for the conservation of wolf on one hand and minimising conflicts with livestock (mainly sheep) production on the other hand. The plan included lethal control of wolves (Box 6.5) "when possible according to international laws, and where necessary" (Marboutin & Duchamp 2005). This plan was followed by the Action Plan 2008–2012 (MEEDDAT & MAP 2008).

In May 2013, the ministers in charge of agriculture and ecology approved the "National Wolf Action Plan 2013-2017", developed by the national wolf group consisting of all relevant interest groups (DREAL 2014a). The plan is in line with the previous plans and defines the principles, the objectives and the means of wolf policy led by the French State for the next four years. It integrates the possibility for adaptation as response to changing situations. The main pillars of this policy are: scientific follow-up of the species, indemnification of damages and support to breeders, measures of a management on a case-by-case basis of the wolf population, communication and consultation, and international cooperation. "*The main goal remains to guarantee the protection of wolves on the French territory while restricting their impact on breeding, the dynamism and diversity of which form a specificity of our country*" (The French official website about wolf 2014[14]).

Box 6.5. Defensive measures against wolf attacks in France according to the Plan d'action national loup 2013-2017 (DREAL 2014a).

In France the use of firearms to prevent wolf attacks is allowed in four different situations:

Tir d'effarouchement: non-lethal shots aimed at scaring the predator. They are allowed and considered an appropriate response for example in case of emergency, when the situation is new and mitigation measures have not yet been put in place to protect the herd.

Tir de défense individuel: lethal shots fired by one individual using a smooth bore gun. They are allowed when all mitigation measures are already in place. The use of rifle gun is allowed only in cases where a herd is protected but has been attacked at least once in the previous two years.

Tir de défense renforcé: In case of repeated attacks on livestock, shooting is allowed by more than one person at a time using rifles. Such authorisation is given for a specific herd based on the history of attacks on that or a neighbouring herd.

Tir de prélèvement: culling of individuals for management purposes.

[14] http://www.rdbrmc-travaux.com/loup/spip.php?article89, Ministère de l'Ecologie, du Développement Durable, des Transports et du Logement & Ministère de l'Agriculture, de l'Alimentation, de la Pêche, de la Ruralité et de l'Aménagement du Territoire.

Italy. The Italian Ministry of Environment with technical support of the Istituto Superiore per la Protezione e la Ricerca Ambientale ISPRA has established National Action Plans for brown bear and wolf in Italy (Anonymous 2012). The wolf plan "*provides the formal Italian policy on the species, which is based on a stringent protection regime, support to damage prevention measures, and full compensation of economic damage*" (Genovesi 2002, Anonymous 2012).

Under the Italian legal framework, the responsibility of wolf conservation is spread over different levels with the Ministry of Environment setting the national conservation policies, with the support of ISPRA. The ISPRA functions in an advisory capacity to the Ministry as well as to the regions and autonomous provinces (P. Genovesi, pers. comm.). The responsibility of the actual implementation of the conservation policies lies within each of the regions or autonomous provinces (P. Molinari, pers. comm.). In particular, regional and provincial laws provide incentives for prevention and compensation for damages to livestock with a great diversity of provisions (e.g. concerning the amount reimbursed and procedures for reimbursement). Several LIFE programs were instrumental to permit the application of these measures.

Any removal of wolves, either through culling or translocation, would require an authorisation from the Ministry of Environment, based on a technical recommendation of ISPRA. No derogation has ever been granted for culling wolves under the national legislation and under article 16 of the Habitat Directive, respectively. A pilot removal of hybrids has been carried out within the LIFE program Ibriwolf. Within that project, a protocol to identify wolf-dog hybrids has been developed, and the removal of hybrids has been authorised by the Ministry of Environment, based on a technical opinion of ISPRA. There are ongoing efforts to apply the same scheme to other geographical contexts of Italy to control the growing threat of hybridisation (P. Genovesi, pers. comm.).

The implementation of the National Action Plan for wolf conservation has been criticised by Boitani & Marucco (2013), who have stressed that the administrative fragmentation is to be considered an important threat to wolf conservation. Under the Italian legal frameworks, national actions plans do not have a formal legal power (Anonymous 2012).

Switzerland. A Concept for the management of wolf was developed in 2004 (BUWAL 2004b) and revised in 2008 and 2010 (BAFU 2008, 2010). The Concept defines rules for livestock damage prevention and the removal of wolves in case of too much damage (Box 6.6). The FOEN is responsible for the overall management of the species. However, for the implementation of the Concept, the cantons are in charge. To harmonise the implementation, the country was divided into 8 management compartments. For each compartment, an inter-cantonal commission coordinates the decision-making (in regard to monitoring, prevention and intervention measures). In June 2012, the revised Hunting Ordinance was enacted and currently, the Wolf Concept is revised as well. The regulation of wolf shall be eased in case of livestock depredation (von Arx & Manz 2013).

Based on the revised hunting ordinance, the wolf management plan was revised with the input of the national Working Group Large Carnivores and published for consultation. The results of the consultation was however extremely diverging and not conclusive at all. As at the same time, a parliamentary initiative demanded an adaptation of the federal hunting law (which will have an impact on the hunting ordinance and consequently on the legal frame for the management plans), the endorsement of the new wolf management was cancelled by the respective minister in charge in fall 2014 until the parliament and the council have decided on the new legal framework for large carnivore management in Switzerland (R. Schnidrig, pers. comm).

Box 6.6. Swiss wolf concept: criteria for removing individual "problem" wolves (BAFU 2010).

- All damages must occur within a pre-defined perimeter (prevention perimeter I: area of regular wolf presence and area where lynx also cause livestock damage; perimeter II: areas adjoining "perimeter I" and areas where damages have occurred during wolf dispersal/expansion
- The wolf must have attacked and eaten at least 35 animals within a period of 4 months or 25 animals in one month
- If damages have been caused over a year, the threshold for the next year is reduced to 15 animals (if protection measures requirements have been fulfilled and if no other measures are feasible)
- The canton can authorise the wolf to be shot by a person having the appropriate permit
- A shooting permission is given for a period of 60 days
- The animal must be shot within a pre-defined area

Germany. Several Länder have developed regional wolf management plans, action plans or guidelines (see below). According to Reinhardt (2013) these plans or guidelines, although called management plans, mainly deal with regional conflict mitigation and management competences. The plans do not define any population goals or management measures acting on the population level. Although a national management plan is not under consideration, a general framework on wolf management exists (BfN 2010; Reinhardt 2013).

Austria. In Austria, each of the seven states in the Alpine region is responsible for nature conservation and game management within the limits of the Habitat Directive (Knauer & Rauer 2013). The KOST (see above) drafted a Wolf Management Plan (KOST 2012) which was finalised in 2012 (Schäfer 2012). According to Kaczensky & Rauer (2013) there are no explicit population goals for wolves in Austria defined, besides acknowledging the legal requirements Austria has to protect immigrating wolves. Knauer & Rauer (2013) consider the management plan as a compromise between stakeholders upon which it was agreed in the meetings but they mention that not every part of the document will be supported in the public by each of the interest groups. Nevertheless, the plan is considered as a valuable basis for future discussions (Knauer & Rauer 2013).

Slovenia. Slovenia has a strategic management plan and the first five-year action plan (Majić Skrbinšek et al. 2011) is currently being implemented. There are no explicit population goals for wolf in Slovenia, however the action plan foresees a continuation of the wolf culling with the following goals: prevention of hybridization with dogs, reduced illegal killings, improved local public acceptance and raising awareness of wolves, maintenance of hunters' support to wolf conservation and keeping wolves wary of humans (Majić Skrbinšek 2013). A revised action plan for the period 2013–2017 will be adopted in 2015 by the Slovenian government (M. Jonozovič, pers. comm.).

There are huge differences between the national management plans in many regards, for instance concerning main goals (e.g. conflict reduction versus species conservation), responsibilities and defined instruments. A policy analysis of the wolf management plans of the Alpine countries would help to find out similarities and discrepancies, which in turn can potentially support or hinder the development of an Alps-wide management of the species. Kaeser & Zimmermann (2012) provided a template for a comparison of management plans that we have adapted for the Alps and that could be useful for such a compilation (Appendix V).

Regional management plans lynx

So far, only Bavaria has a regional lynx management plan (StMUGV 2008).

Regional management plans Wolf

Regional management plans for wolf exist in Germany and Switzerland. Nine German Länder developed a regional wolf management plan (for an overview see Reinhardt et al. 2013). As stated above, these plans or guidelines, although called management plans, mainly deal with regional conflict mitigation and management competences. The plans do not define any population goals or management measures acting on the population level (Reinhardt 2013). For the Alpine wolf population, only the Bavarian management plan is of relevance (StMUGV 2007, Bayerisches Landesamt für Umwelt 2014b).

The following Swiss cantons – all are within the Alps – have implemented their own management plans: Grisons (Kantonale Arbeitsgruppe "Grossraubtiere" 1999), Bern (Volkswirtschaftsdirektion Bern 2007), Uri (Amt für Forst und Jagd 2008), Lucerne (Amt für Landwirtschaft und Wald 2009), Nidwalden (Amt für Justiz 2009), Obwalden (Amt für Wald und Raumentwicklung 2009), Fribourg (Service des forêts et de la faune 2010), Schwyz (Amt für Natur, Jagd und Fischerei 2010), and St. Gallen (Volkswirtschaftsdepartement St. Gallen 2013). Similar to the German regional management plans, the Swiss plans also mainly define regional conflict mitigation measures and management competences.

6.2. Hunting and wildlife management practices in the Alpine countries

Red deer and roe deer are the most widely distributed ungulates across Europe and the Alpine range; along with wild boar they compose the most important game species (Linnell & Zachos 2011; Chapter 5.3). These populations recovered from a net decline in the 19th and 20th centuries due to widespread unregulated hunting (Putman 2011). Management practises such as regulated and selective hunting practises, increasing migratory corridors and habitat connectivity, reduction in habitat fragmentation and protection of habitat, but also reintroductions, reinforcements and artificial feeding have led to an increase and recently stabilisation of these populations. In many regions of Europe, wild ungulates are so abundant today that management practises include measures to reduce damage to crops and forests and prevention or mitigation of diseases. Hunting is the most important management practise and is used in many countries to control populations and hence limit damage to agriculture and forests (Putman 2011). Culling of wild ungulates is widespread across Europe and is largely linked to the claims of agriculture, forestry and transport sectors (Morellet et al. 2011).

In spite of these challenges, few countries have established robust long-term census system to monitor ungulate populations. Direct and indirect censuses are the most commonly used methods to monitor ungulate populations. Direct census methods may include capture-mark-recapture method (Switzerland), open hill counts (Switzerland), animal vocalisations (Italy), spot lighting (Italy, Switzerland) and drive counts (Italy, Switzerland) (Morellet et al. 2011). Estimates from indirect methods use faecal samples, animal vital rates (France), snow tracking (Switzerland) and habitat quality (France, Slovenia) among several other sampling methods (Morellet et al. 2011).

6.2.1. Hunting systems and wildlife management practices

Wildlife in the Alpine countries is managed through legal and practical means such as protective laws and selective hunting. Legislation operates at different levels (national, regional, provincial, etc.) across Europe. One generality however, exists across European countries: game does generally not belong to the land owner (Putman 2011). Game belongs to everyone or no one – *res communis* or *res nullius*. In the case of *res communis*, the state can either sell hunting licenses or allocate the sale of hunting licenses to individuals or hunting groups and do not involve landowners in this aspect (e.g. Italy, Slovenia[15], Switzerland). In the case of *res nullius*, hunting rights belong to the landowner who allocates licenses while the state has the right to determine management goals (e.g. Austria, Germany, France; Table 6.4; Putman 2011).

Although hunting seasons in European countries should ideally be determined based on the ecology and natural history of the species that are hunted, it is currently not the case in several countries (Apollonio et al. 2011). Factors that should ideally be taken into account when determining a hunting season include the period of rut, pre-parturition and post-parturition. These are important factors as hunting during these key moments can disrupt reproduction and have a negative impact on the population. Hunting during periods of late pregnancy can also be negatively perceived by the non-hunters with regard to ethical concerns. Culling adult females with young can result in the death or loss of fitness of young animals still dependant on their mothers. Many European countries allow the hunting of animals during these three critical periods during the breeding season for species such as red deer, roe deer, chamois and wild boar (Apollonio et al. 2011). Table 6.5 gives an overview of hunting dates for key ungulates across the Alpine countries (these dates are only indicative and are partly adjusted at a local or regional level).

Table 6.4. Comparison of management systems across the Alpine countries (adapted from Putman 2011), showing strong state controlled management practises on the left and individual landowner management types on the right.

	Impose/determined by state (National or regional authorities)	**Proposed by land owners associations/ Hunters' associations, approved by State**	**Proposed by landowners associations/ Hunters' associations or equivalent voluntary**
Game management district/group	Switzerland, Slovenia, France, Austria	Germany, Italy	
Management objectives	Switzerland, Slovenia, France, Austria	Germany, Italy, Austria	
Management Plan	Switzerland, Slovenia, France	Germany, Italy, Austria	
Quota/Cull Targets	Switzerland, Slovenia, France	Germany, Italy, Austria	
	Cull carried out by game wardens	**Individual licenses allocated (per animal)**	**Global quota allocated to leaseholders**
Global Quota/ Individual licenses	Switzerland (Canton of Geneva), France	Switzerland, France	Slovenia, Germany, Italy, Austria

[15] In Slovenia, the state is the legal owner of game according to the Environmental Protection Act of 2004.

Table 6.5. Comparison of open season for hunting ungulates in the Alpine arc [m= male, f= female, s= sub-adult]. These dates may vary locally and regionally and especially in countries like Austria where there is no country-wide law, or there may be additional distinctions for age classes. Sources: Apollonio et al. 2011; IT : Raganella Pelliccioni et al. 2013; SL: M. Jonozovič, pers. comm.

	Red deer	**Roe deer**	**Chamois**	**Wild boar**
France	23.08–28.02 01.03–31.03 (coursing) [m/f/s]	15.05–31.08 (stalking), 01.09–28.02 (driving), 01.03–31.03 (coursing) [m/f/s]	23.08–28.02 [m/f]	15.04–14.08 (stalking), 15.08–28.02 (driving), 01.03–31.03 (coursing) [m/f/s]
Italy	15.10–15.12 [m/f]	01.06–15.07 and 15.08–15.11 [m], 15.09–31.12 [f]	01.08–31.12 [m], 01.09–31.12 [f]	15.04–31.06 (stalking) [m/s], 01.10–31.06 (stalking) [f], 01.11–31.06 (driving) [m/f/s]
Switzerland	01.08–31.12 [m/f]	01.05–31.01 [m/f]	01.08–31.12 [m/f]	01.07–31.01 [m/f/s]
Liechtenstein	01.05–30.11	01.05–30.11	01.08–31.12	01.05–30.11
Germany	01.08–31.01 [m/f], 01.06–31.01 [s]	01.05–31.01 [m/f]	01.08–15.12 [m/f]	15.06–31.01 [m/f/s]
Austria	01.05–31.01 [m/f]	01.05–31.12 [m/f]	01.06–31.12 [m/f]	All year [except for sows with piglets]
Slovenia	01.07–31.01 [m/f]	01.05–31.10 [m], 01.09–31.12 [f]	01.08.-31.12 [m/f]	All year [m/s] 01.07–31.01 [f]

France. Ungulate populations at the beginning of the 20th century in France were limited in their expansion due to overharvesting (Chapter 5.3). After 1979, hunting quotas and selective shooting were imposed and ungulate translocations were carried out to strengthen the populations in different regions across the country (Maillard et al. 2010)

Wildlife and environmental monitoring are carried out by the *Office National de la Chasse et de la Faune Sauvage* ONCFS. Each department has a Hunter's Federation (*Fédération Départementale des Chasseurs* FDC) which is a group of hunters that are in charge of activities related to wildlife management (e.g. fixing hunting quotas for game species each year) and their health as well as protecting their habitats (FACE 2007). Hunting quotas are generally based on population densities and economic value of the forest. The final quotas are subject to approval by the government. The importance of the economic value of a forest is related to the damage caused by wild ungulates when browsing. In France, hunting rights can also be attributed based on property rights, therefore a hunter must either own the hunting territory or be a member of an Approved Communal Hunting Associations ACCA. In theory, all landowners in departments which have the ACCA system automatically belong to one of these associations. The departments are divided into management units which are monitored by the FDC. There are two main hunting laws In France: the "Hunting Law" (26th July 2000) and the "Law on the development of rural territories" (23rd February 2005). Hunting plans have been required for all deer since 1979 and for chamois since 1992.

Future hunters are also trained by the FDC on the practical and theoretical aspects of hunting. Further education and courses for landowners and hunters are also carried out by the FDC. The FDC activities not only include the management of game stocks but also the control of damage caused by ungulates and management of ungulate population sizes (FACE 2007).

Italy. The role of hunting in Italy is primarily to control wild boar, red deer and roe deer populations (Apollonio et al. 2010). The national hunting law 157/92 controls the main aspects of ungulate hunting in Italy and all hunting plans need to be evaluated by ISPRA, which oversees the application of the open seasons. A significant part of the daily administrative regulation is controlled by the provinces and districts (*Ambito territoriale di Caccia* ATC [General Hunting Districts]). Italy has five different types of land organisation with regard to hunting practises which include (1) single general hunting districts, (2) ATC/CA hunting districts, (3) ATC/CA with sub-hunting districts and to types of municipal hunting reserves, namely (4) ATC – General hunting districts, and (5) CA – Alpine districts. Hunting rights are based on these land organisation types. For example, in some areas hunters are only allowed to hunt in the district they belong to and in the Eastern Alps, which mainly consists of municipal reserves, only citizens of the municipality are allowed to hunt there (Apollonio et al. 2010). The national hunting law allows the hunting of chamois, roe deer, red deer and wild boar as well as other species which are not relevant to our report. Wild boar are hunted during three months of the year (October–January). The durations and months are subject to change by the Regional and Provincial Governments.

Switzerland. The current Federal Hunting Law came into force in 1986 with its main goals being the maintenance of biodiversity, protection of threatened species (e.g. ibex), limiting the damage caused by free-ranging animals, and a sustainable hunting of wildlife populations (Imesch-Bebié et al. 2010). There are 41 federal wildlife reserves where hunting is banned, created after the first Federal Hunting Law of 1875, with the purpose of protecting game species and increasing their populations. There are three different types of hunting systems in Switzerland (Fig 6.1; Nahrath 2000).

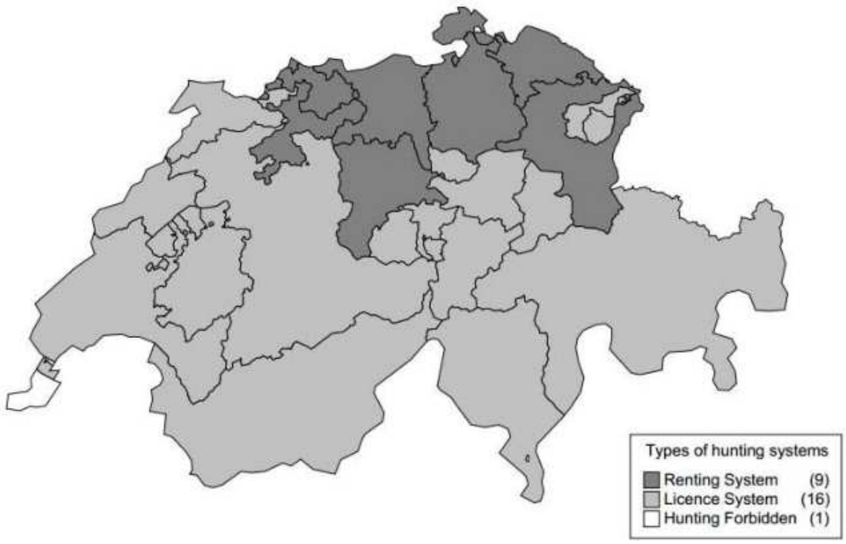


Fig. 6.1. The geographical distribution of the three different hunting systems in Switzerland (Nahrath 2000) North-eastern Switzerland has a renting system where hunters' associations lease a certain area (e.g. the territory of a municipality) from the state, most cantons have a licence system where hunters can use the entire open areas to get their quotas, the Canton of Geneva has no hunting.

Hunting laws and management plans for species that can be hunted are enacted at the cantonal level. There are two types of hunting administrative systems: one based on annual licences ("Patentjagd" or license-based system) and the other based on land which is leased for hunting purposes ("Revierjagd" or rent-based system; Nahrath 2000). Wildlife management units differ in size based on the canton and the system of hunting practised. In the case of the licence system, the territory can be divided into smaller wildlife management units. All hunters require a licence which can be obtained following a course and exam (practical and theoretical) which provide them with knowledge about wildlife, rifle handling and shooting. The licence system is practised in 16 of the 26 cantons. In the case of the license system, the state hunting commission meets once a year to determine the beginning of the hunting season, the duration of the hunting period for the different species, the price of licenses, the quotas for each hunter, etc. These decisions are taken based on wildlife statistics which are collected mainly by state game wardens. Nine other cantons practise the land-leasing system which allows a land to be leased for a duration of eight years and for a limited number of hunters. This system works on the basis of associations which rent parcels of hunting land which belong to different communes. The hunting association is responsible for the management of the rented land, wildlife monitoring and the hunters. The canton of Geneva has prohibited hunting since 1974 and ungulate management in this canton is carried out by state game wardens.

Germany. Ungulate management and hunting practises in Germany are carried out with the objective of reducing and preventing damage to crops and forests. The basic wildlife legislation is almost the same as the "Reichsjagdgesetz" which was put into place in 1931 except for a major change with regard to the federal hunting law, wherein the 16 states now have different regulations (Wotschikowsky 2010). These differences include variations in the length of the hunting season, minimum size for the "Revier", hunting rights and landownership, leasing rights, annual harvest plans and hunting rules. A single "Revier" is the smallest ungulate management unit and may vary between 75 ha to 1,000 ha (2,000 ha in mountains). Annual harvest plans need to be approved by the regional authorities who also collect harvest reports from the various "Reviere". Annual shooting plans are required for all species except for the wild boar. Hunting grounds can also be collectively organised into a "Hegegemeinschaft" HG for species such as the red deer which are mobile and move over large territories. These HG can be as large as 200-500 km^2 and in some cases even bigger. The corresponding landowners manage the species and the hunting plans in these HG's.

Austria. Austria uses the "Reviersystem" similar to the system in Germany; the Austrian "Bundesländer" are responsible for managing the hunting laws across the country. The country is divided into nine provinces and each province has a separate hunting law to manage its wild ungulates (Reimoser & Reimoser 2010). Each province is further divided into hunting districts which correspond to wildlife management units. There is no country-wide law. Hunting rights are controlled by landowners and are an important source of income. The nine hunting laws from the different provinces define the species which are allowed to be hunted, wildlife-damage compensation, duration of shooting season (they vary between the provinces) and some provinces have the additional possibility of creating habitat-protection areas. The overall aim of all nine hunting laws are to maintain a high wild species diversity, protect wildlife populations, avoid damage of vegetation by wildlife, and promote the sustainable use of wild animals.

A number of legal provisions regulate hunting and these include hunting acts of several "Bundesländer" (and include the corresponding enforcement regulations), Land Nature and Conservation Acts and Regulations listing protected fauna and flora, Land Animal Protection Acts, Land Environment Protection Acts, Federal Forestry Act, Federal Weapons Act, Federal Animal Disease Act and others (FACE 2002). Every hunting permit holder is a member of at least one "Landesjagdverband" (hunting association).

Slovenia. Game in Slovenia is owned by the state according to the Environmental Protection Act of 2004 (Adamic & Jerina 2010). The current Law on Wildlife and Hunting was adopted in 2004 and controls the wildlife management system in Slovenia (Adamic & Jerina 2010). In the same year, hunting seasons were fixed for various species by the "Act on Hunting Seasons and Wildlife". The country is divided into a number of protected areas which comprise of 15 Game Management Districts, 413

Game Management Areas/hunting grounds and 12 State Hunting Grounds with Special Purpose and together they occupy about 90% of the country (M. Jonozovič, pers. comm.). The Slovenia Forest Service prepares ten-year wildlife management plans for each of the Wildlife Management Areas. Harvest quotas are determined on a yearly basis by the Slovenia Forest Service in collaboration with the hunters, official nature conservationists, landowners, farmer's organisations and foresters (M. Jonozovič, pers. comm.). The hunting quota and duration for adult wild boar and yearlings are flexible as this species is responsible for a high percentage of damage to crops. There are also special permissions for extended shooting seasons to control red deer populations due to extensive damage to agriculture.

The "Core Slovene register of large game species and large carnivores" was established in 2004 and has since then maintained detailed records of animal mortality (including hunting, road accidents and epizooties; Adamic & Jerina 2010).

6.2.2. Conservation issues and problems

Predation and predation impact on wildlife and livestock are described in Chapter 5.5.

Pathogens and epizooties

Lynx. Disease and infections have been reported to cause only 3.5–25% of mortality cases in lynx, while 54–96.5% of the known mortality cases were caused by human-related activities (Ryser-Degiorgis 2009). In 1999, five lynx were found suffering from mange (*Sarcoptes scabiei* and *Notoedres cati*) in Switzerland (Ryser-Degiorgis et al. 2002). An inter-specific transmission of mange can occur between foxes *Vulpes vulpes* and lynx (Ryser-Degiorgis et al. 2002). However, due to the relatively solitary behaviour of lynx, the risk of disease transmission is relatively lower than in other species (Ryser-Degiorgis 2009). In another study based on 72 samples, 18% of lynx mortalities were attributed to infectious diseases, some of which could have been transmitted by domestic animals (Schmidt-Posthaus et al. 2002). The rate of infected animals is thought to be higher as 15 animals that were found were radio-collared (Schmidt-Posthaus et al. 2002). When taking only radio-collared animals into account, the percentage of animals that died due to infectious diseases rose to 40% (Schmidt-Posthaus et al. 2002). Therefore it is likely that the percentage of animals dying due to disease is higher than what is known, as not all carcasses are found and examined. A case of mortality of a lynx kitten was thought to have been caused by gastrointestinal parasites (Ascarids; Schmidt-Posthaus et al. 2002).

Wolf. Epizooties can be a hindering factor in the survival and expansion of populations of wild species which are recolonising a historical range. Wolves generally host a number of parasites which are harmless as such, but when combined with factors such as malnutrition or bacterial/viral diseases, they can become harmful (Kreeger 2003). The IUCN/SSC Canid Specialist Group considers the grey wolf to be susceptible to sarcoptic mange, canine parvovirus, distemper and rabies (Mech & Boitani 2004). Viral diseases such as canine distemper virus (CDV), infectious canine hepatitis and rabies are present across the world. However, rabies, a potentially dangerous zoonosis, is currently not a problem in the Alpine region (www.who-rabies-bulletin.org). In specific conditions, pathogens can have a negative impact on wolf population increase and expansion especially in areas where transient individuals are common such as in the French Alps (Kreeger 2003). Bacterial diseases such as brucellosis are common in ungulates and carnivores around the world and can also affect wolf pup survival (Kreeger 2003). No cases of epizooties have been reported from the Alps. Outside the Alpine range, helminthic fauna (intestinal parasites) were studied in a population of wolves in Latvia and 17 species were found in 34 individuals (Bagrade et al. 2009). In Croatia, a case of visceral leishmaniosis was discovered in 2003 (Beck et al. 2008). During a study carried out between 1986 and 2001, 92 dead wolves were recorded in Croatia of which five (5.4%) were infected with rabies (Huber et al. 2002). In 2013, the general health of the grey wolf in Slovenia was thought to be good (Žele & Vengušt 2013).

Wild ungulates. Wild ungulates have been known to be the host and reservoirs of many common infectious diseases (Artois 2003). Multi-host diseases are of particular concern for the conservation and management of wildlife populations (Gortazar et al. 2007). Wildlife abundance and open air livestock husbandry are some ways that allow the transmission of diseases from wild ungulates to domestic livestock (Gortazar et al. 2007). This can have a high economic impact for livestock owners (Ferroglio et al. 2011). For example, brucellosis is a bacterium which can affect wild and domestic ruminants and in some cases even humans (Ferroglio et al. 2011). It can be transmitted when infected wild ungulates come in contact with domestic cattle. Mycobacterial infections such as Bovine tuberculosis (bTB, *Mycobacterium bovis*) in red deer populations in France and paratuberculosis (*M. avium paratuberculosis*) which affected the Italian red deer population are some diseases of concern in the wildlife-livestock interface (Gortazar et al. 2007, Ferroglio et al. 2011). Sarcoptic mange (*Sarcoptes scabiei*) and keratoconjuntivitis (*Mycoplasma conjunctivae*) affect chamois, domestic sheep and goat and are of particular concern as they affect population numbers. These diseases were recently spreading within the Alpine ungulate populations and across Europe (Giacometti et al. 1998, Gortazar et al. 2007). Managing domestic livestock and controlling their movement to reduce exposure with infected wild ungulates is an important way of reducing the risk of spreading diseases from wild to domestic animals. Therefore ungulate control and management is also linked to economic factors and health risks. Controlling these diseases requires a good knowledge of diseases present and their distribution as well as the general health of domestic and wild populations which they can affect (Ferroglio et al. 2011). Wildlife diseases in Europe are currently managed at a regional, national and international level (Ferroglio et al. 2011). Disease monitoring, prevention of introduction of a disease and control of existing diseases are some management schemes (Ferroglio et al. 2011). A more difficult scheme is the eradication of a disease; however, this is an expensive method and almost impossible to achieve (Gortazar et al. 2007, Ferroglio et al. 2011). Measures to protect livestock herds from large carnivore attacks will also allow for a better observation and surveillance of domestic ungulates with regard to infectious diseases.

Other conservation issues and challenges

Variable legal status and administrative units. The different hunting legislations and legal status of carnivores and ungulates across these countries are a challenge for large-scale wildlife conservation (Chapter 6.1). In federal countries, the coordination at national level is already a challenge, and e.g. in Austria and in Italy, the cooperation between the national subunits with regard to wildlife management is weak. International coordination and cooperation will hence be even more difficult.

Inconsistencies in management system within and between countries. Throughout the Alpine region, there is a variety of systems to manage ungulate populations as well as predators and domestic livestock. Depending on national and regional traditions and legislation, hunting, and especially harvest of wild ungulates, can have a strongly diverting local societal and economic significance. These differences are very strong between the western and eastern Alps, e.g. illustrated by the annual harvest of game (Chapter 5.3). While it would be an illusion to harmonise wildlife management and hunting systems across the Alps, it is at least important to understand the differences and to consider them with regard to the integration of large carnivores into the existing wildlife management traditions.

Lack of reliable wild ungulate data. The only data that is consistently available on wild ungulates are hunting bags, and not even this information was available for most of the administrative units from the Italian Alps (Chapter 5.3). Some of these data may be collected, but they are not shared, let alone made publicly available. The really big challenge of reintegrating large carnivores into the modern landscapes will be their impact on wild prey species and hence the conflict with hunters. This challenge will be difficult to address as long as there are no reliable data on wild ungulate populations available.

Livestock husbandry practises. Different husbandry practises are used across the Alpine region. In some areas, livestock husbandry has lost its economic importance while it remains an important activity in others (Chapter 5.4). The national policies (including subsidies) vary considerably and go often beyond the economic significance especially of sheep husbandry, as e.g. landscape management and general regional economic goals are included. However, the approach regarding livestock damage control seems to be quite similar in all Alpine countries (Chapter 6.3), based on (1) damage prevention (protective measures), (2) damage compensation, and (3) removal of notorious stock raiders. The third point is the one where the positions are rather different, with Switzerland being the country with the most interventional system and Italy the country with the strictest protective approach.

6.3. Prevention and compensation of predation on livestock

6.3.1 General aspects of compensation and prevention

Compensation. In all Alpine countries losses of livestock to large carnivores are reimbursed by the government or associations (e.g. hunting). A compensation system has been adopted in France, Italy, Switzerland, Germany, Austria and Slovenia. The compensation techniques vary in the different countries depending on the socio-economic status of the country as well as culture and traditional practises (Boitani et al. 2010). In the Alpine countries, most of the compensations are monetary in nature. Except for the case of some provinces in Austria, this compensation is part of a pre-arranged government programme. These programmes include the examination of the dead domestic animal and determination of cause of death by an expert. The “typical” case of lynx depredation is rather easy to identify, whereas it is more difficult to distinguish between attacks of wolves or of stray dogs (Molinari et al. 2000, Fico et al. 2005). If confirmed that the animal was attacked and killed by a lynx or a wolf, the farmer or livestock owner is entitled to a predetermined sum of money which is generally based on the breed and age of the animal. In some countries, the amount of money reimbursed is also based on the proper implementation of anti-predator methods such as livestock guarding dogs, electric fences, night-time enclosures, presence of shepherds etc.

The rational for the compensation is that the legal protection and the recovery of the large carnivores are a societal desire, and that therefore society (hence the state) should pay for losses of those who economically suffer from the return of these animals. However, reimbursement of losses alone is an inadequate measure to solve the conflict. All countries also support the implementation of protective measures, and some countries (e.g. Slovenia, Switzerland and France) allow the lethal removal of "problem animals" (notorious stock raiders) under certain conditions.

Mitigation. With the disappearance of large carnivores from their historical range, the traditional livestock protection methods were also abandoned. It was a common practise in the past, when predators were rare, to leave large herds of livestock unattended in the mountain pastures, in countries like France, Switzerland and the Alpine region of Slovenia. However, the return of large carnivores, in particular wolves, requires a return to traditional pastoral ways and guarding which can be an expensive option. Guard dogs are a relatively effective method of reducing and preventing depredation by dissuading attacks. A study in North America found fladry (hanging flags from ropes) to be an effective method to dissuade wolves from attacking and can be used alongside other preventive measures (Musiani et al. 2003). Non-electric fences do not appear to have an effect on depredation intensity (Mattielo et al. 2012).

The most effective protective measures against predation include guarding dogs, electric fences and the presence of a shepherd. Livestock protection dogs LPDs are a popular protective measure which was used over many centuries and originated in central Europe and parts of Asia (Gehring et al. 2010). Other methods include the regrouping of sheep and putting them in enclosures in the evening (Ministère de l'écologie, du développement durable, des transports et du logement 2014). With the disappearance of predators in the Alps, the use of such dogs was abandoned. The return of wolves and lynx to regions where livestock grazes has prompted conservation GOs and NGOs to encourage livestock owners to use traditional methods to reduce the risk of losing animals and reducing the economic impact of livestock depredation. The dogs defend the sheep herds against attacks by wolves and stray dogs (Chevallier et al. 1999, Gehring et al. 2010). The presence of LPDs also reduces the likelihood of wild ungulates grazing in the same pastures as sheep thereby reducing the risk of disease transmission (Gehring et al. 2010). However, aggressive LPDs can interfere with unrelated human activities such as hiking and tourism (Gehring et al. 2010).

LPDs are used in various parts of the Swiss Alps (Mettler & Lüthi 2008). There are currently about 200 LPDs working in Switzerland (AGRIDEA 2014a). The number of LPDs in the department Alpes de Haute-Provence, France, increased from 21 in 2004 to 244 in 2009 (Anonymous 2010).

LPDs are not the only measure used to protect livestock. An overview of measures is given in Table 6.6. Fladry is used mostly as an emergency, highly mobile measure and exclusively to dissuade wolves. It works for a limited duration until they are accustomed to such defensive measures. The presence of experienced shepherds is a crucial aspect for other preventive measures to be successful.

Table 6.6. Cost, maintenance and effectiveness of various livestock protection measures (Gehring et al. 2010).

Measure	**Cost**	**Maintenance**	**Effectiveness**
Woven wire fence	High	Low	High
High tensile electric fence	Moderate	Moderate	Moderate-High
(Electrified) Fladry	Low-moderate	Moderate-High	Low-moderate
Frightening devices	Low	High	Low
Repellents	Low	High	Low
Livestock protection dogs	Low	Moderate	High

In addition to direct predation, flocks of sheep can also be victims of indirect fatalities caused by panic due to the presence of predators. There have been cases of large numbers of sheep running randomly and falling off cliffs and hill sides having been provoked by panic. Such occurrences have been recorded across the French Alps and can lead to the loss of over 80 sheep resulting in economic difficulties for the sheep owners (Duchamp et al. 2004).

6.3.2. Prevention and compensation of predation of lynx on livestock per country

France. Depredation cases of lynx on livestock in the French Alps are low (Chapter 5.5.1). The average number of depredation cases attributed to lynx in the whole of France between 2000 and 2011 was 72 and the compensation cost amounted to less than 20,000 € per year (Marboutin 2013b).

Switzerland. In 2013, there were 21 cases of depredation in the Alps. The compensation of losses of sheep and goat between 2006 and 2011 amounted to 6,500–25,000 CHF per year (Chapter 5.5.1). Killed livestock have to be examined by an official and trained person and are compensated up to 100% if predated by lynx.

The Federal Office for the Environment FOEN pays 80% of the amount; the rest is paid by the canton concerned. Three cantons (Solothurn, St. Gallen, Zürich) make payments to hunting societies for having lynx in their hunting ground (von Arx & Zimmermann 2013), to compensate for the reduced hunting bag.

Preventive measures are only implemented when lynx repeatedly attacks the same herd; then the measures would be the same as against wolves (see below). If a lynx kills more than 15 sheep within a given area per year, the canton can ask for permission to remove the individual (Box 6.4, Chapter 6.1.2). The last time that such a case was registered was in 2003 (Fig. 5.41; Chapter 5.5.1), indicating that livestock depredation by lynx is not a major cause of conflict at the moment. More important than livestock predation by lynx is the conflict with hunters over reduced game availability (roe deer and chamois).

Germany. Currently, only the states of Bavaria and Lower Saxony have provisions for compensation payments for cases of lynx depredation, as the other "Länder" have not had any damages yet. All kills have to be assessed and documented by trained personnel.

The main conflict over lynx in Germany is with hunters with regard to the predation of wild ungulates. A "reporting premium" is paid by hunter associations of Bavaria and Lower Saxony to hunters for reporting confirmed lynx kills of wild ungulates (Kaczensky et al. 2013b).

Austria. In the Austrian "lynx areas", along the Austrian-Bohemian-Bavarian border (outside the Alps) and around the Kalkalpen NP, sheep farming is a relatively unimportant activity. In recent years, there have been no cases of livestock depredation in this region. In the case of damages, a "voluntary" (no legal base for compensation) reimbursement would be available in most provinces and is covered by the hunting insurance of hunting associations.

The predation of wild ungulates is the main source of the conflict between lynx and hunters in Austria. To encourage reporting, a "reporting premium" is paid by the hunters associations of the province of Upper Austria for confirmed lynx kills of wild ungulates (Kaczensky et al. 2013c).

Slovenia. All lynx depredation cases are compensated by the government (Kos & Potočnik 2013). Compensation costs varied between 1995 and 2014 from 137–13,225 € (M. Jonozovič, pers. comm.; Chapter 5.5.1). In some areas, livestock are brought back to stables or fenced in at night to reduce the risk of attacks in the Slovenian Alps (AGRIDEA 2014b).

6.3.3. Prevention and compensation of predation of wolf on livestock per country

France. The attacks began in 1993 and increased steadily until 2005 (Fig. 5.49; Chapter 5.5.2). On average 10 to 15% of flocks in the wolf range are attacked each year. Of those attacked, 70% of the flocks are attacked only once, while only less than 10% are repeatedly attacked more than five times (up to 20–30 times) (Chapter 5.5.2). After each attack, whether by lynx, wolf or dog/other, a damage assessment is carried out if possible within the first 48 hours of the attack (DREAL 2014b). The characteristics of the attack, state of the victim are recorded and the cause of attack are determined.

Predation "hot-spots" have been identified when studying sheep numbers and duration of exposure to the predation risk (Marboutin 2013a). When examining wolf presence and the locations of the attacks, such hot-spots are clearly visible (Fig. 6.2).

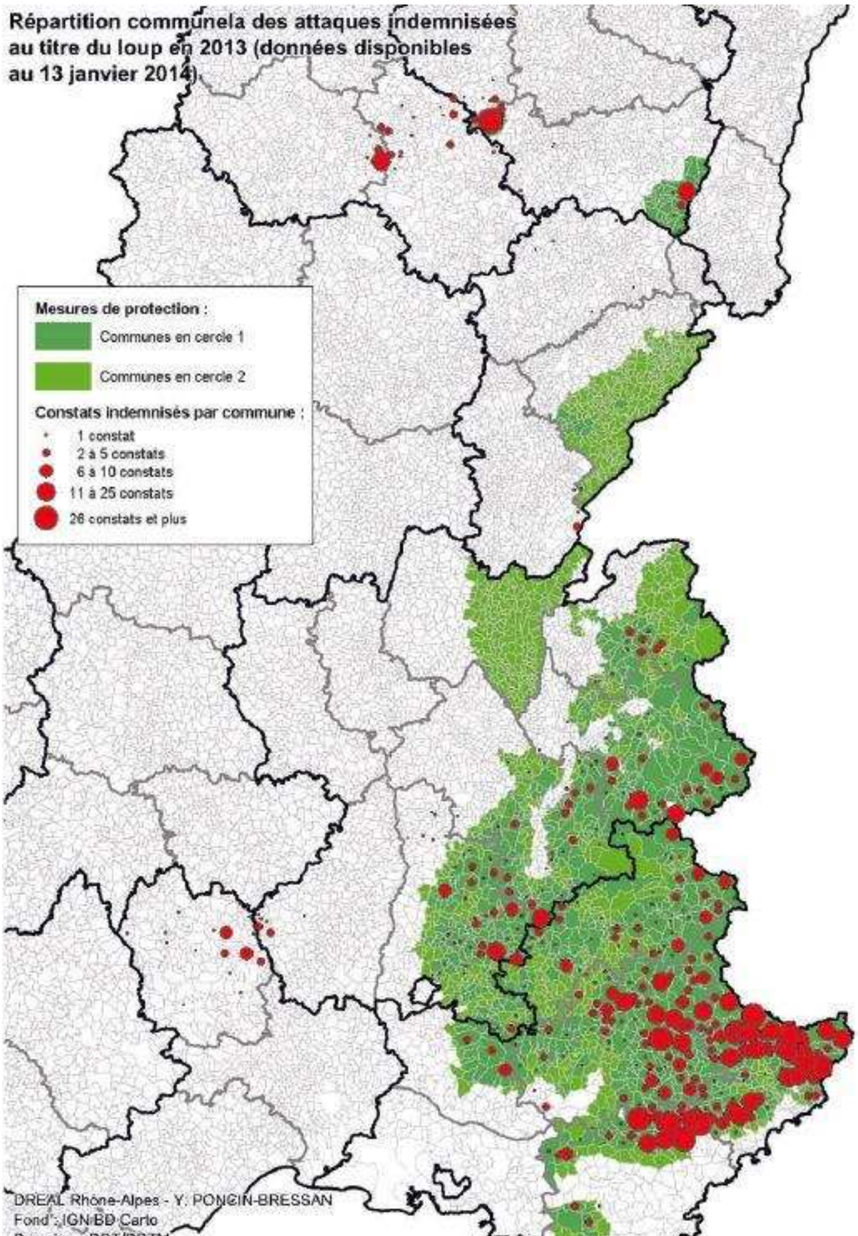


Fig. 6.2. Compensation of domestic livestock attacks by wolves per commune in the year 2013. The size of the red dots represents the number of attacks: 1, 2-5, 6-10, 11-25 and more than 26 (ONCFS 2014a).

As is the case in most Alpine countries, the greatest number of attacks occurred during the summer season when livestock graze on alpine pastures (Fig. 6.3; Anonymous 2010).

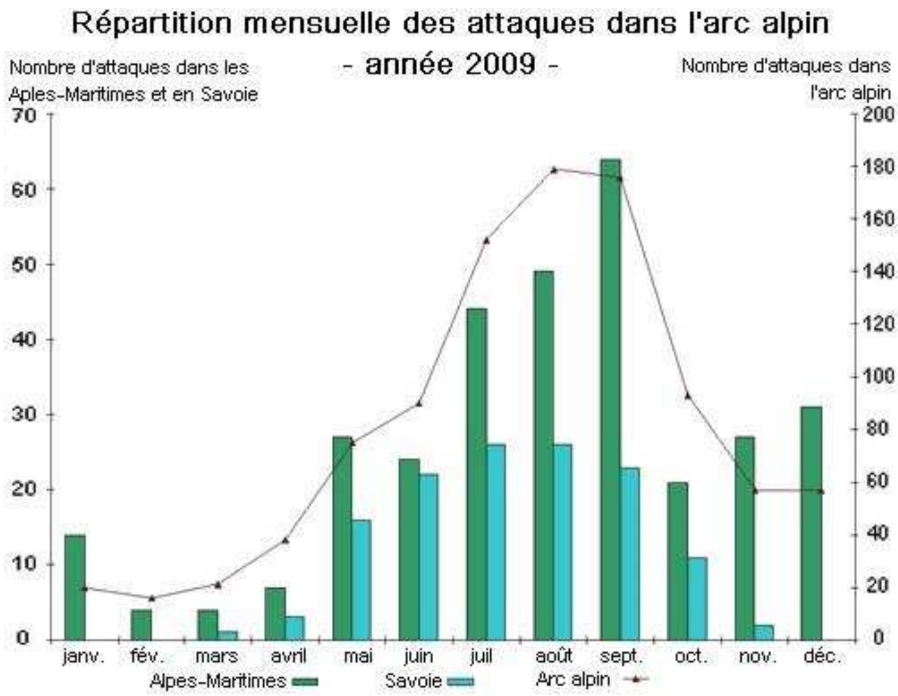


Fig. 6.3 Distributions of wolf attacks in the departments of Alpes-Maritimes, Savoie and in the Alpine Arc, per month (Anonymous 2010).

In France, compensations are paid for three cases: direct losses, animals missing and indirect losses (DREAL 2014a). The compensations paid for wolf damage in all of France increased from 0.79 million € in 2008 to 2.3 million € in 2014 (DREAL 2014a, DREAL 2015). During this period 85% of the predation cases reported received compensations. Wolf attacks and compensations paid in the nine Alpine departments for the years 2012–2014 are listed in Table 6.7.

Table 6.7. Number of victims of wolf attacks for which compensations were paid, and amount of compensations paid in French Alps between 2012 and 2014 (DREAL 2015).

Department	Number of victims			Compensation €		
	2012	2013	2014	2012	2013	2014
Alpes de Haute Provence	1,000	910	1,225	309,283	315,433	402,267
Alpes Maritimes	2,417	2,449	2,731	763,397	786,921	864,830
Drôme	218	366	188	75,785	111,272	57,895
Hautes Savoie	179	78	12	63,295	18,291	4,341
Hautes Alpes	514	441	661	158,074	143,187	232,531
Isère	203	111	1,010	58,101	40,149	234,805
Savoie	453	404	567	138,195	130,556	157,943
Var	712	804	1,070	265,379	302,525	364,497
Vaucluse	36	22	20	9,484	9,022	8,716
Total French Alps	**5732**	**5585**	**7484**	**1,840,993**	**1,857,356**	**2,327,825**

Compensations cover direct losses based on a pre-arranged scale taking into account injuries and death. Preventive measures are strongly encouraged and supported by the government. In 2014, 12 million € were spent on prevention measures (J. Transy, pers. comm.). These include: large guarding dogs (about 1,200 dogs in the Alps), subsidised extra-herding (shepherds), and electric fences. Livestock protection dogs and night fencing schemes have shown to reduce the risk of predation (Anonymous 2010). Unguarded sheep flocks suffer about a dozen attacks on average but when using appropriate protective measures, the number of attacks was reduced and resulted in only 1–3 victims. Preventive measures also include the removal of individual wolves which are known to cause damage. The wolves can be shot based on a number of criteria defined in the national action plan (Box 6.5; DREAL 2014a).

Italy: In 2011, there were 383 cases of livestock damage mostly on sheep and goat in Piedmont. Wolf attacks on domestic livestock were found to be significantly higher during the months of May to October (Fico et al. 1993, Gazzola et al. 2005; Chapter 5.5.2). During these months, livestock can be found in Alpine meadows and may receive little or no protective measures to reduce the possibility of attacks by predators (Fico et al. 1993). Wolf depredation cases were also investigated in the northern Apennines and found to be lower than in other regions but focused in certain areas/farms (Reggioni et al. 2005). The verification process of livestock depredation is often done by personnel who are not trained to carry out such assessments (Fico et al. 2005). As free roaming dogs are also responsible for livestock depredation, there is a risk of an over estimation of wolf predation on livestock when assessments are made by untrained people.

Claims are made to the local forest department within the first 24 h and a standardised procedure to assess the damage is carried out. In the Piedmont region for example, the assessment is carried out by a veterinarian (Dalmasso et al. 2012). Livestock owners are compensated for all injuries and damages to livestock by both wolf and dog unless in cases where the dog can be located and the attack positively identified. In the Piedmont region, the total cost of direct losses was 68,000 € in 2010 and 72,953 € in 2011. An additional 19,703 € were spent for indirect losses (Dalmasso et al. 2012, Boitani & Marucco 2013).

In the Alps, damages are limited thanks to intensive damage prevention programmes implemented by the Piedmont region and by providing electric fences, guarding dogs and veterinary support to shepherds and farmers. There appears to be a relative decrease in the number of attacks or victims despite the increase in the number of wolf packs (Fig. 5.51; Chapter 5.5.2). Italy uses a number of protective measures including livestock protection dogs, fences and shepherds. In the Piedmont region, although depredation is mostly caused by wolves, a great number of cases can also be attributed to stray dogs (Tropini 2005).

The administrations at the regional level in Italy use a variety of approaches to manage livestock predation conflicts. These methods are different in the various provinces and range from no intervention at all to allocation of funds for prevention methods (guarding dogs, electric fences, salary for shepherd, etc.) and also to financial incentives to subscribe to insurance policies against predation by wolves. However specific information regarding methods used in each province is not available. There is no national database of livestock damage and many regional administrative departments do not keep formal records of the compensation schemes, making it impossible to have a comprehensive figure on damages at a national level (Boitani et al. 2010).

Switzerland. In the Swiss Alps, 114,000 CHF for 280 animals killed by wolves was paid in 2011, and 48,500 CHF for 135 animals in 2012 (KORA 2014).

Livestock kills have to be examined by an official person (state game warden in cantons with licence hunting, designated and trained person in cantons with renting hunting system) and losses are compensated to 100% if wolves are found to be the cause of death.

The Federal Office for the Environment FOEN pays 80% of the amount, while 20% are paid by the canton.

Electric fencing, livestock guarding dogs and shepherds are the main prevention methods used to protect livestock in Switzerland. Livestock owners are financially supported by the FOEN. The canton can ask for permission to remove individual problem wolves in cases where the criteria defined in the Swiss Wolf Concept are met (Box 6.6, Chapter 6.1.2). Between 2006 and 2011, four such wolves were legally shot.

LPDs are increasingly being used in areas with wolf presence in Switzerland (Fig. 6.4).

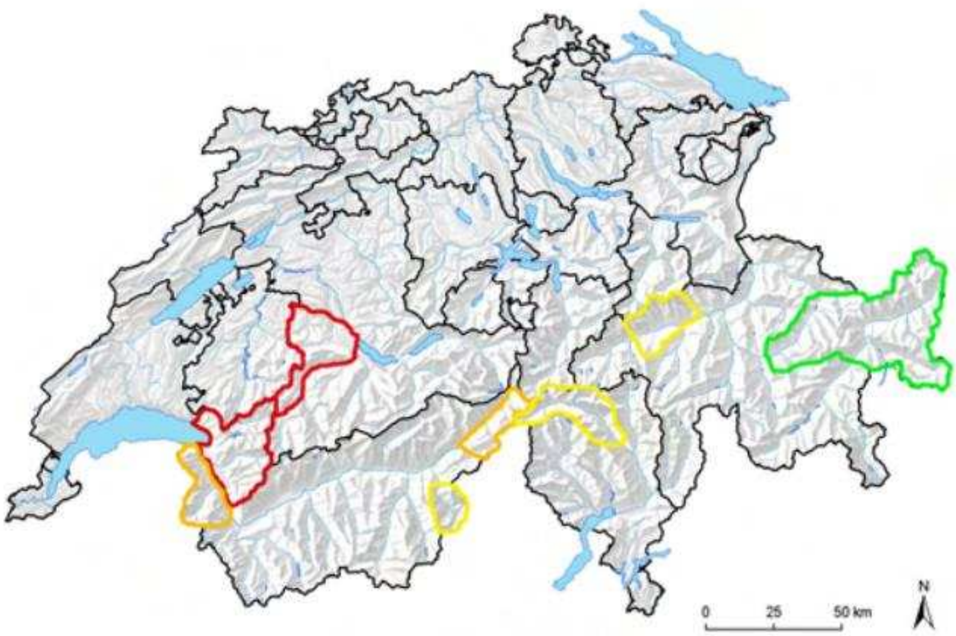


Fig. 6.4. Priority areas for livestock protection measures in Switzerland in 2007 (coloured polygons). Most areas are concerned with wolf attacks, but the green polygon demarcates the region where bears are occasionally immigrating. The priority areas are regularly reconsidered. (Source: Mettler & Lüthi 2008).

Germany. Wolf-livestock conflict in Germany is currently low compared to other European countries presumably due to the low wolf presence in the region (Reinhardt et al. 2010, 2012). In 2011, a total of 26,584 € were paid in compensation for 225 domestic animals killed by wolves all over Germany (Reinhardt 2013; for Alps see Chapter 5.5.2). While livestock protection measures like electric fencing are relatively easily applicable in the German lowlands, such measures are more difficult to use in the Alps.

Austria. Between 2009 and 2011, there were 15-70 cases of livestock damage. Compensations are only paid when livestock mortality is confirmed to have been caused by predators, however actual amounts paid are not available. These payments are “voluntary” as there is no legal right to compensation, and in most provinces they are covered by the hunting insurance of hunting associations. Compensation payments do not cover additional labour costs or the burden of proving that the animal was killed by a predator. These responsibilities lie with the livestock owner.

With the relatively rapid expansion of the wolf population across the Alps, a pilot project for livestock protection on alpine pastures was launched in 2012. The programme included the testing of the efficiency of fencing, herding, livestock guarding dogs etc. in five pilot areas (the programme was launched only in one area at the time and results not made available; Kaczensky & Rauer 2013).

Slovenia. Damages caused by wolf have started to occur in the Slovenian Alps in 2006 and are thought to be possibly caused by a single wolf. Up to 26 animals are killed per year and annual damage compensations amount up to 3,869 € (M. Jonozovič, pers. comm.). Damages to livestock are systematically compensated by the government, but the compensation system has been criticised as it does not encourage breeders to invest in efficient protective measures to mitigate attacks by predator.

The majority of the attacks are restricted to a very small number of livestock breeders with very poor protective measures against large carnivores, however actual data is unavailable. To be eligible for compensation, the livestock breeder must meet certain legal requirements of livestock protection. However, these requirements are so low that meeting them does not provide effective protection against large carnivore attacks. Initiatives to change the compensation system and correct these weaknesses are being put in place (Majić Skrbinšek 2013).

6.4. Discussion and conclusions

Legislation. Successful wildlife conservation requires a good scientific understanding of biological/ecological as well as human dimension aspects, a solid legal framework and efficient wildlife management institutions to implement the measures needed to conserve even species such as large carnivores that are sources of conflict.

Wolf and lynx are strictly protected by international and national laws, but with regard to practical management, almost all countries having substantial populations of these carnivores are applying some regulations allowing for exceptional removals of problem animals. Both Switzerland and France have e.g. removed lynx that were notorious stock raiders (Table 6.2). For the wolf, France is applying the principles of “*tir de défense*” and “*tir de prélévemènt*”, and Switzerland has set limits for how many livestock a wolf is allowed to kill before it can be lethally removed (Table 6.3). Not only for the Alpine countries, but across Europe it is today a much debated question how much deviation from the strict protection international regulations such as the EU Habitat Directive or the Bern Convention would allow and tolerate.

Wildlife management plans. The national and regional management plans in the different countries vary as their aims and objectives are quite different. For example, emphasis can be given to species conservation or to managing conflicts. Furthermore, the legal embedding (and hence the legal obligingness) of national action plans can differ. Some were drafted by expert groups, some were developed in a participatory approach and some went through a wide consultation process. The WISO Platform of the Alpine Convention would now call for a tighter cooperation of the Alpine countries, what would imply a certain synchronisation of the national/regional action plans. In such a process, the “Guidelines for the population level management plans for large carnivores” (Linnell et al. 2008) is providing guidance and a template (Box 6.2) for such a process.

Compensation and mitigation. The return of the large carnivores puts a lot of pressure on the sheep husbandry system as it has been established over the past 50 years. Although losses to large carnivores are financially compensated and preventive measures supported, the habit of letting sheep graze free on alpine and subalpine pastures is simply no longer possible with the presence of wolves. This requires a substantial change of the husbandry system with the respective personnel and leading to financial consequences. Until now, the financial loss caused by large carnivores was insignificant compared to the amount of financial support invested into livestock husbandry, but it is obvious from all statements of sheep owners and their organisations that financial compensation alone is not the final solution of the conflict between sheep breeders and the wolves. The challenge for sheep husbandry, mountain agriculture, and the society as a whole is rather to cope with the new situation and the necessity to change a system that is considered a traditional way of livestock husbandry, although it has developed only over the past 50–100 years.

7. Assessment of the future development of the lynx and wolf populations in the Alps

The recolonisation of previously occupied habitat and the expansion of a recovering species or population are determined by factors such as the habitat and landscape features, land-tenure system, dispersal characteristics, resource availability and distribution, as well as human attitudes and activities (Zimmermann 2004). The dispersal of carnivores such as lynx and wolf depends mainly on resource availability (e.g. prey), landscape (habitat and topography), and the social structure of the species (Zimmermann 2004, Zimmermann et al. 2007). Anthropogenic pressure is often believed to be the main factor limiting the distribution of large predators, which are generally not very habitat dependent (Zimmermann 2004).

7.1. Recolonisation by wolf and lynx: Similarities and differences

The most obvious difference between the renaissances of the two large carnivores is that wolves recolonise the Alps spontaneously, whereas lynx were reintroduced (Chapter 3). Lynx from the Carpathian Mountains have been released in different parts of the Alps since the early 1970s (Breitenmoser & Breitenmoser-Würsten 2008). Of seven attempts to establish local populations within the Alps, only three were successful. However, most of the releases involved only very few individuals. As far as known, no lynx from an autochthonous population has ever made it to the Alps, although the western Carpathians and the eastern Alps are less than 100 km apart. Wolfs however return to the Alps spontaneously from several autochthonous populations in southern or eastern Europe. They started to recolonise the south-western Alps in the early 1990s as a consequence of the recovery and the spread of the population in the Apennine (Fabbri et al. 2007, Marucco 2009, Caniglia et al. 2012). The most important Alpine population established in the south-western Alps in France and Italy. However, wolves from the north-eastern European population and from the Dinaric population have also immigrated to the Alps (Zedrosser 1996, Rauer et al. 2013, WAG 2014). At least in one case, a female from the Italian population and a male from the Dinaric population have reproduced (see below; SloWolf 2012). The unlike history of the recolonisation highlight the specific biological features of the two species regarding dispersal, social and land-tenure system, and expansion capacity. These differences have consequences for the dynamics of the recolonisation and for the (genetic) viability of the newly established populations, which must be considered in the respective conservation and management approaches.

Habitat

The primary habitat for lynx and wolf in Europe is forest, but they are not restricted to it. The lynx has a preference for forest and shrub habitat and, compared to wolf or brown bear, is the species with the most specific demands with regard to habitat and prey base (Zimmermann 2004, Becker 2013). In central and western Europe, lynx are clearly linked to forested habitat and their distribution overlaps more or less the distribution of forests in Europe (Zimmermann 2004, Breitenmoser & Breitenmoser-Würsten 2008). Suitable forested areas must be well-connected and have a certain extent to provide sufficient space for cover and for travelling (Rüdisser 2001). However, although lynx are a forest-dependent species, they are able to use other habitat types as long as enough prey and cover (vegetation, broken terrain) to stalk prey are available (Rüdisser & Martys 2002).

Meanwhile, wolves are a very adaptable species and are habitat generalists (Boitani 2000, Fuller et al. 2003). They are not primarily a forest species and can live in different habitats, wherever sufficient food is available and where they are tolerated by humans (Box 7.1.; Boitani 2000, Chapron et al. 2003, Fechter & Storch 2014).

Box 7.1. Habitat types occupied by wolves in Eurasia (Fechter & Storch 2014, Ministère de l'Ecologie, de l'Energie, du Développement durable et de l'Aménagement du territoire & Ministère de l'Agriculture et de la Pêche, no date).

- In France, wolves are currently mainly present in forests of mountain areas;
- in Italy and Romania, some wolves occupy shrub land and areas close to garbage dumps but are also found in forested areas;
- in Poland, wolves occur mainly in meadows and wetlands beside their primary habitat forest;
- in Portugal, wolf occurrence is associated with livestock abundance;
- in Russia, wolves inhabit mainly mosaic habitats of forest and agricultural landscape;
- in Spain, wolves use agricultural landscapes and pine plantations;
- in Switzerland, the first wolf pack established itself in a forested mountain area.

The differences in habitat use are possibly associated with the coexistence with humans and the availability of prey in different regions (Jędrzejewski et al 2008, Fechter & Storch 2014). Nevertheless, wolves show a certain preference for forest in most parts of their range (Marucco 2009, Fechter & Storch 2014). In Europe in general, large forest areas are particularly suitable for wolves (Boitani 2000). Such areas offer cover, day resting sites, suitable habitat for wild prey, less disturbed habitats for denning and rendezvous sites (Massolo & Meriggi 1998, Boitani 2000). Also, during the ongoing recolonisation process of the Alpine Arc, wolves seem to primarily use forested areas and establish their territory there (Ciucci et al. 1997, Boitani 2000). High quality habitat for wolves were defined to be areas with a high proportion of forest cover, low human impact and high wild prey abundance (Boitani 2000, Jędrzejewski et al. 2008, Marucco 2009). The three key variables associated with the presence of wolves and habitat suitability were road density, human population density and forest cover (Ciucci et al. 1997, Jędrzejewski et al. 2008, Fechter & Storch 2014).

In the Alps, the slope seems to be an important criterion with regard to habitat selection for lynx (or rather the variable that encompasses most of the other positive and excludes negative factors influencing lynx presence). In Switzerland, most of the day resting sites and breeding places were found on steep slopes, for which lynx seem to have a certain preference (Rüdisser 2001). It was thought that the gradient is a good indicator for potential disturbance by humans or dogs (Rüdisser 2001). Moreover, lynx like to stay close to windfall areas and clearings, because roe deer often stay close to such structures (Rüdisser 2001). Lynx tend to avoid areas of permanent human activities such as settlements or intensive agricultural areas (Zimmermann 2004). However, when lynx occur in good quality habitats, they can adapt to human presence and semi-natural landscape regardless of their disturbances (Zimmermann & Breitenmoser 2007). Several observations of lynx indicated that they did not particularly care about human activities such as recreation, logging, hunting, etc. In Switzerland, for example, radio-collared lynx were often located in a military exercise site when shooting was going on and on several occasions, lynx were located close to logging places, next to a mountain restaurant, ski lifts or recreational areas (Zimmermann 2004). One collared lynx, called Turo, which was translocated to the eastern Swiss Alps, stayed over one year close to the city of Zurich before he returned to the north-eastern Swiss Alps (Breitenmoser & Breitenmoser-Würsten 2008).

Land-tenure system

Four factors influence the land-tenure system of lynx: presence of conspecifics, prey availability, expansion of suitable habitat and the topography. Lynx are living in stable individual territories and are solitary except for females with their offspring of the year (Breitenmoser & Breitenmoser-Würsten 2008).

Exclusive home ranges are essential for adult Eurasian lynx for reproduction (Schadt 2002) and are larger for males than females (Molinari-Jobin et al. 2010). Generally, adult males share their home ranges with one or two females, but home ranges do not or only marginally overlap intrasexually (Kramer-Schadt et al. 2005, Molinari-Jobin et al. 2010). The size of the range can vary highly; Signer (2010) found them to be between 60–480 km^2 for females and 90–760 km^2 for males (see also Appendix III). A radio-telemetry study in the Alps, revealed mean home ranges of male lynx to be 137 km^2 (range 74–199 km^2, n = 11, 95% Kernel) and of females 76 km^2 (range 45–164 km^2, n = 12, 95% Kernel; Breitenmoser-Würsten et al. 2001). Home range size varied depending on habitat type, composition of prey community and prey density (Molinari-Jobin et al. 2010). Main roads, large rivers, or high mountain chains often operate as home range borders (Schadt et al. 2002a). Although lynx are solitary animals, they are also social and need regular contact with their conspecifics. These contacts are however indirect (e.g. scent marks) and do only occasionally involve direct meetings. Lynx tend to establish home ranges adjacent to those of other lynx (Zimmermann 2004, Zimmermann et al. 2007, Becker 2013). Lynx are only able to establish a permanent home range if there is free space; otherwise, they will be pushed to less suitable habitats (Zimmermann 2004). In Switzerland, subadults tended to establish a home range close to the home ranges of adult resident lynx (Zimmermann 2004). Females stayed closer to their natal range than males. Indeed, young females can even take over their mothers territories (Zimmermann et al. 2005, Breitenmoser & Breitenmoser-Würsten 2008). This specific land-tenure system affects expansion of a population or the establishment of new populations: a lynx population expands along its rim, but has difficulties to overcome (as a population!) broad barriers. At the same time, it is not very likely that isolated dispersing lynx can establish new population nuclei disconnected from already settled areas (Becker 2013), although long-range dispersals were observed also in the Alps (Fig. 7.1).

Wolves are territorial, too, but are structured in highly social packs, which consist in general of a dominant single-breeding pair and their offspring of several generations (Marucco & McIntire 2010, Marescot et al. 2012). Mean wolf pack size in the Western Italian Alps was estimated at 4.4 during a 10-year intensive study (Marucco & McIntire 2010). In northern Italy, pack size was on average 5.6 ± 2.4, and in Poland, packs included 4 to 6 animals with an average size of 4.9 (Krutal & Rigg 2008, Caniglia et al. 2014). The pack can be considered as the social functional unit of wolf populations (Chapron et al. no date), and the basic social unit is formed by the mated dominant pair (Mech & Boitani 2003). Their behaviour influences density, home range configuration, reproduction and mortality (Marucco & McIntire 2010).

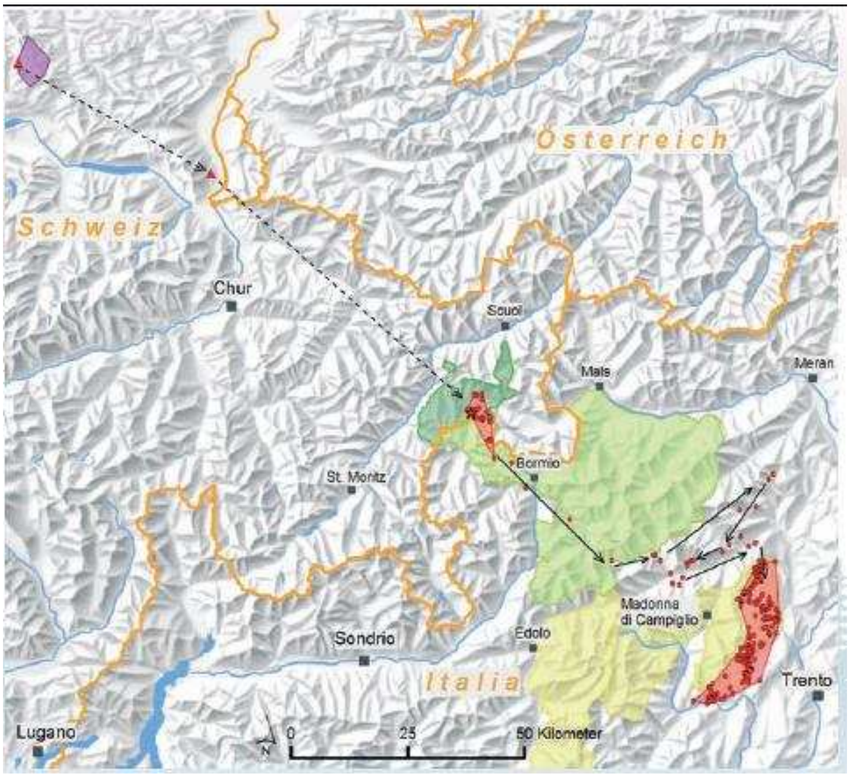


Fig. 7.1. Dispersal of the young lynx B132 from eastern Switzerland to Trentino in Italy (Haller 2009). Star: Point of capture. Circles: GPS-locations. Triangles: Camera trap images. Violet area: Maternal home range. Red areas: Areas of longer stays. Dark green area: Swiss National Park. Light green area: Parco Nazionale dello Stelvio. Yellow area: Parco Naturale Adamello/ Ademello Brenta.

Home ranges of wolves depend on wolf pack size, density and population status as well as prey species abundance (Fechter & Storch 2014). Wolves live at low densities over large territories of about 300 km^2 in Europe (WAG 2014; see also Appendix III). In the Slovak Carpathians, the home range of a radio-tracked male wolf living in a pack was 146 km^2 (100% Minimum Convex Polygon MCP) and 191 km^2 (100% MCP) for a female from another pack (Findo & Chovancová 2004). The average home range size of wolf packs in Europe varies from 80 up to 2,000 km^2 (Ministerium für Umwelt, Gesundheit und Verbraucherschutz des Landes Brandenburg 2009). However, home ranges of individual wolves and packs studied by radio-tracking in Europe (42°-53° N), varied in size only between 87 and 243 km^2 (Findo & Chovancová 2004). In the Lausitz, Germany, the mean wolf pack home range size was estimated at 215 km^2 (n = 12; Fechter & Storch 2014). The average home range of a wolf pack in the Apennines was estimated at 197 km^2 (MCP) based on radio tracking (Ciucci et al. 1997). The size of wolf pack territories in the Piedmont region (Italy) were on average 101 km^2 (based on scat collection and MCP) with a maximum territory size of 213 km^2 (Marucco & Avanzinelli 2012). In the western Alps the minimum territory size of wolves lied between 50 to 300 km^2 (Marucco et al. 2009) and wolf pack home ranges in the Alpine regions were estimated to vary between 200 – 400 km^2 (Herrmann 2011).

Dispersal

Dispersal is an important aspect for the colonisation of new areas (Zimmermann 2004). It has an essential influence on the persistence and dynamics of populations and the distribution of species (Zimmermann et al. 2005). Understanding dispersal patterns is necessary for the conservation and management of species in human-dominated landscapes (Zimmermann 2004). Moreover, dispersal plays an important role in regard to the survival of spatially structured populations (Schadt 2002).

Lynx observed in the Swiss studies had a moderate tendency of large scale dispersal and do not tend to disperse very far (for an exception see Fig. 7.1). Lynx are highly mobile animals and can move through unfavourable habitat and cross barriers such as highways or rivers, but they disperse slower and less far than wolves (Zimmermann 2004, Zimmermann et al. 2007, Becker 2013). Obstacles such as high mountain peaks, glaciers, major highways, large rivers, lakes and settlements may be overcome by individuals, but they are barriers to the expansion of the population (Rüdisser 2001, Zimmermann 2004, Becker 2013).

In Switzerland (incl. Jura Mountains), the mean dispersal distance of lynx was 39 km (Zimmermann 2004). Dispersal distances are generally larger for males than for females (Zimmermann 2004, Becker 2013). The dispersal distance in the Swiss Alps averaged 31 km (5–56 km) for males and 19 km (7–33 km) for females (Breitenmoser & Breitenmoser-Würsten 2008). The maximum known dispersal distance in the Alps was 200 km straight line from a lynx born in the eastern Swiss Alps which dispersed to the Italian Trentino (Fig. 7.1; Haller 2009). In Poland, dispersal distances of lynx ranged from 5 to 129 km with the maximum distance reached by a male (Zimmermann 2004).

Mortality rate of subadult lynx during dispersal is high at about 50% (Breitenmoser & Breitenmoser-Würsten 2008). Preliminary observations of lynx dispersal in the Alps suggested that both dispersal rate and distance could be negatively density dependent (Zimmermann 2004, Becker 2013). Mean dispersal distance in a high-density population was 25.9 km, and 63.1 km in the low-density population (Zimmermann et al 2005). There is no convincing explanation for this observation so far, but it may infer that high density situations (“peaks” in local lynx populations with erupting conflicts as discussed in Chapter 5.5.1) may not lead to further expansion of the population.

Compared to lynx, wolves have a higher dispersal potential and can disperse over longer distances (Valière et al. 2003, Fabbri et al. 2014, WAG 2014). Usually, when young wolves reach complete sexual maturity at 1–3 years of age, they begin to disperse (Marescot et al. 2012). However, wolves can disperse at a wide variety of ages from 5 months to up to 5 years old (Mech & Boitani 2003). Any wolf (of both sexes) disperses from its natal pack unless it gets a breeding position which happens seldom (Mech & Boitani 2003, Kojola et al. 2006).

Wolves show different types of dispersal; some establish a territory next to the natal one, some move around the local populations and others travel over hundreds of kilometres (Mech & Boitani 2003). Sometimes wolves first only temporarily leave the pack several times before leaving it definitely and some individuals may disperse and settle twice or more (Mech & Boitani 2003). There are indications that dispersing wolves maximize breeding opportunities rather than resource acquisition. Therefore, they may travel long distances due to the low probability of finding a mate (Ciucci et al. 2009). Dispersing wolves appear to have a higher tolerance for unsuitable habitat and to be more mobile than a breeding pack (Herrmann 2011). During dispersal, wolves can move over large areas of unsuitable or poor quality habitats, but successful establishment of a territory or formation of a pack is limited to large areas of high quality habitats (Marucco 2009, Marucco 2011, Falcucci et al. 2013).

Dispersal rate between females and males seems to differ. In Finland, where wolves immigrated from the Karelian population, in the first years almost only males arrived (Zedrosser 1996). During recolonisation of the Alps from the Apennines, migration was male-biased. All inferred first-generation migrants to the Alps were males (Fabbri et al. 2007). In the ongoing recolonisation of the Swiss and Austrian Alps, mainly males appeared (Fig. 7.2, 7.3).

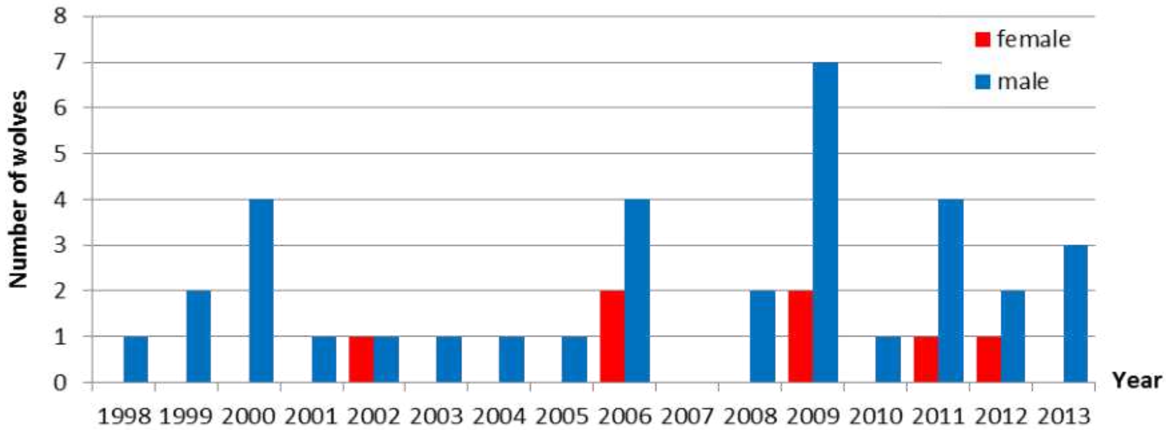


Fig. 7.2. First records of individual wolves dispersed into Switzerland. The number and sex of wolves recorded the first time in Switzerland are displayed per year (R. Manz, KORA, pers. comm.).

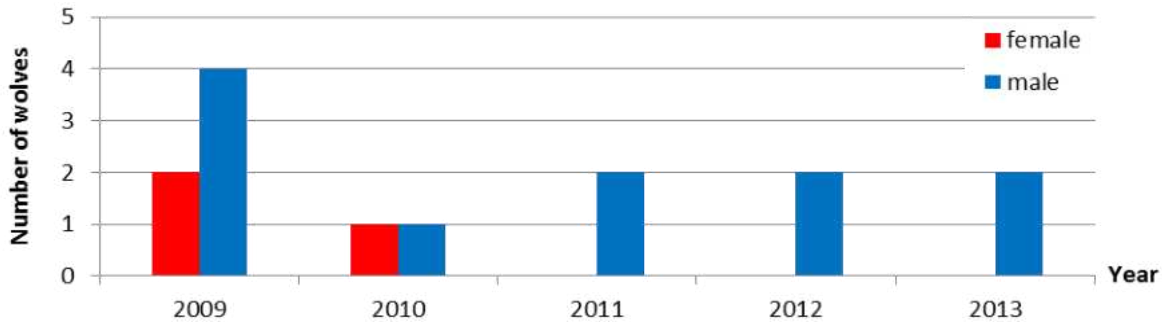


Fig. 7.3. First records of individual wolves dispersed into Austria. The number and sex of wolves recorded the first time in Austria are displayed per year. Wolf monitoring started in 2009 in Austria (G. Rauer, pers. comm.).

Box 7.2. Long-distance dispersal events of wolves from and to the Alpine countries.

- In 1994, a wolf from the Tatras Mountains located in the Czech Republic, around 800 km away, dispersed to the Bavarian forest (Germany) where it was killed. Another individual from the same region was recorded in Austria in 1996 (Landry 1996).
- In 2009, a male wolf dispersed from Germany across Poland to Belarus over 1200 km in 86 days (Ministerium für Umwelt, Gesundheit und Verbraucherschutz des Landes Brandenburg 2009).
- A male dispersed over 520 km from the Ligurian Alps in Italy to Germany where he was detected in Bavaria (Marucco & Avanzinelli 2012).
- A wolf from Italy, which was recorded in the French Alps in 2008 and was then detected in the canton of Grisons (Switzerland), moved through the northern Tyrol (Austria) and was finally identified in Bavaria (Germany) in 2009 (WAG 2014).
- Another wolf dispersed from the Western Alps to the northern Apennines in Italy travelling at least 958 km in 10 months (Fabbri et al. 2014).
- The GPS-collared male wolf, called Slavc, born in Slovenia in 2010, dispersed from the Dinaric population in 2011 to Italy (SloWolf 2012, Fabbri et al. 2014). He was first recorded in the Trentino province before he established a territory in the Veneto region in the Lessinia Regional Park in Italy in 2012, where he reproduced with a female from the Italian wolf population (Fig. 7.4, SloWolf 2012, WAG 2014).

There seem to be no significant differences between male and female dispersal distance (Mech & Boitani 2003, Kojola et al. 2006). Several long-distance dispersal events of wolf over 1,000 km have been recorded so far. In North America, wolves were documented to have dispersed over more than 1,224 km distance and the maximum distance travelled per day was 20 km (Fabbri et al. 2014). Most long-distance dispersers (over 300 km) were males (Fabbri et al. 2014). In Finland, mean dispersal distance was around 100 km, ranging from 35 to 445 km (Fabbri et al. 2014), while average dispersion distance from the Piedmont Region (Italy) was 92.6 km ± 106.3 km and from the northern Apennine Mountains (Italy) 52.97 ± 40.17 km (Marucco & Avanzinelli 2012, Caniglia et al. 2014). Long-distance dispersal has also been reported from and to the Alpine countries (Box 7.2; Landry 1996, Fabbri et al. 2014; WAG 2014).

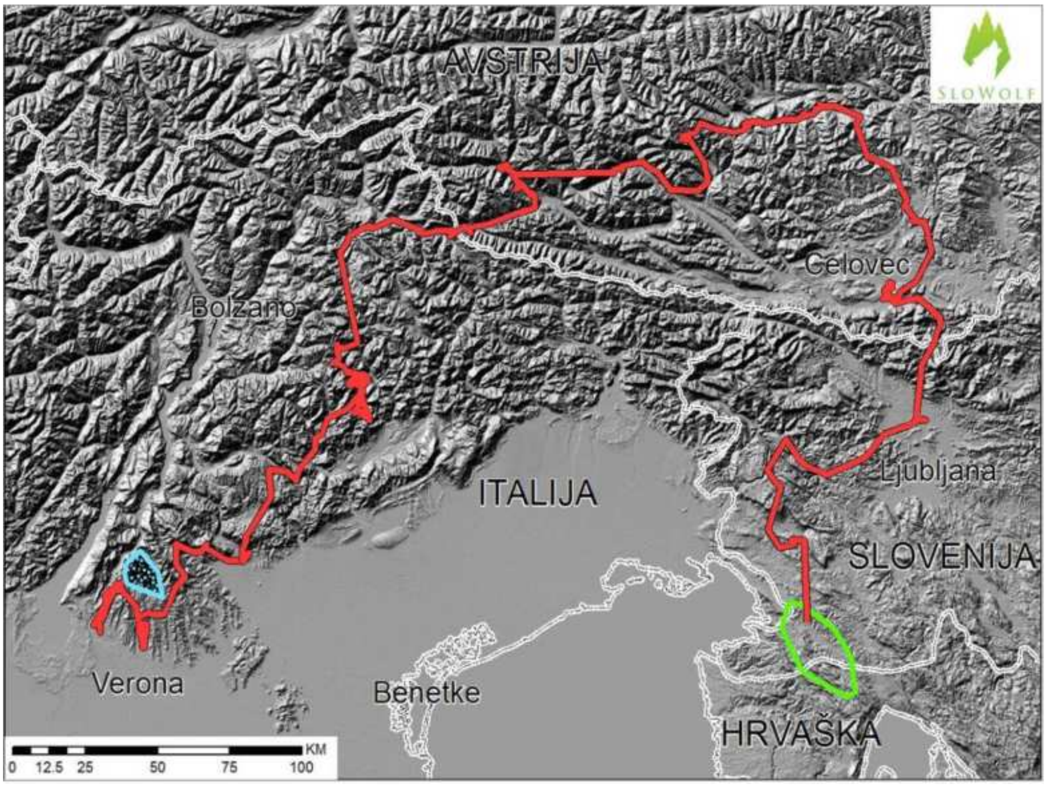


Fig. 7.4. Dispersion of a male wolf, Slavc, from Slovenia to Italy; blue = new established home range, green= home range before dispersal (SloWolf 2012).

7.2. Potential distribution, abundance and expansion dynamics of the lynx in the Alps

A number of habitat suitability models addressing the lynx over the whole Alpine range and parts of the Alps or adjacent areas exist (Appendix VI). Habitat predicted as suitable for lynx varies depending on data input and model techniques. In the following subchapters, we present the different models and their outputs but we do not discuss in detail the methodological approach or the validation of the models as this would go beyond the scope of this report.

7.2.1. Habitat suitability and potential distribution

Alpine range

As far as we know, only three different lynx habitat suitability models for the entire Alpine range have been published. The habitat model of Zimmermann (2004) based on VHF telemetry data from the Swiss Alps and the Jura Mountains and using Ecological Niche Factor Analysis ENFA as an analytic tool, predicted suitable lynx habitat distributed over the entire Alpine Convention area: a total area of 93,579 km^2 was indicated as suitable lynx habitat (Fig. 7.5). Larger areas of habitat with low suitability were predicted in the north-western and central French Alps, in the central Italian Alps, in parts of south-western and central Switzerland, the northern part of the German Alps, parts of the eastern, south-eastern and central Austrian Alps and bits of the Slovenian Alps (Zimmermann 2004). Most of these low suitable areas lie in regions with very high elevations.

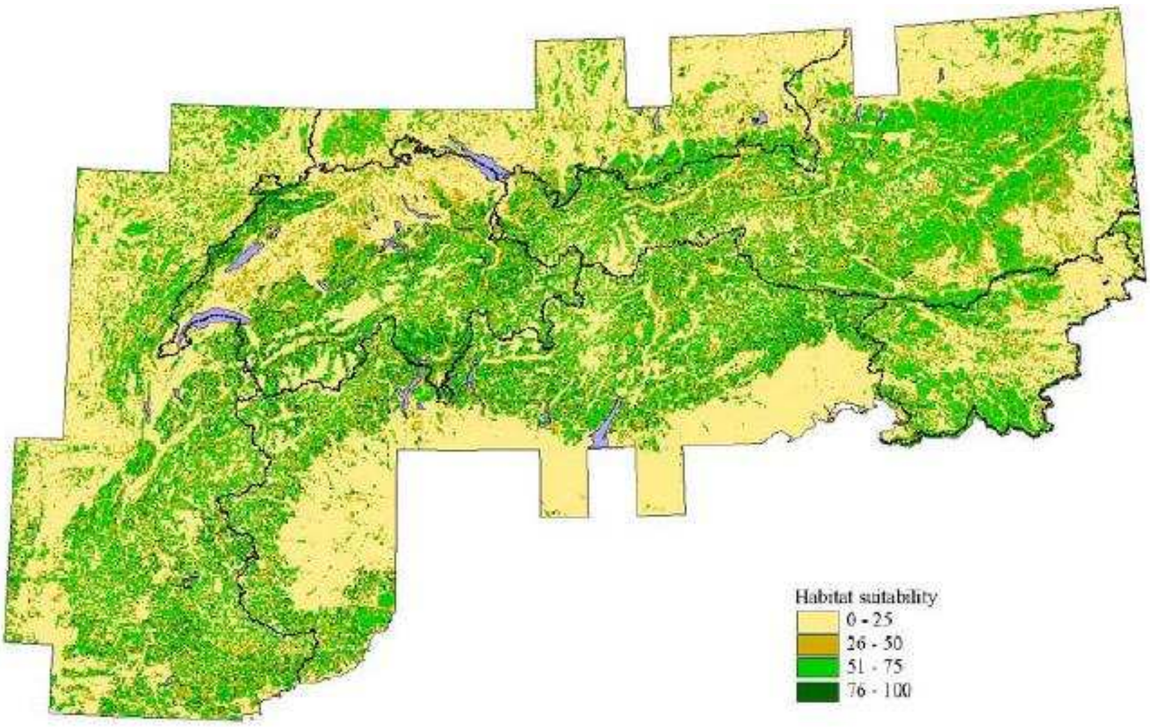


Fig. 7.5. Map of predicted lynx distribution based on Ecological Niche Factor Analysis ENFA. Habitat suitability index: the higher the value the more suitable is the habitat (Zimmermann 2004).

Lynx preferred forests, shrubs and herbaceous vegetation and avoided areas of intensive agriculture. Lynx presence was also negatively correlated to the frequency of urban areas presuming that lynx tend to evade areas of high human activity (Zimmermann 2004). However, areas used by lynx are not necessarily free of human activity and presence. Distance to roads was not negatively correlated to lynx presence in most of the models of Zimmermann (2004), indicating that, when lynx occur in good habitats, they can adapt to human presence.

The habitat suitability model of Signer (2010), based on a logistic regression with chance observations from Austria as input data, predicted suitable lynx habitat distributed all over the Alpine region (Fig. 7.6). Very high altitude areas were considered to be unsuitable for lynx. Besides these regions, larger areas of low suitability were indicated in the south-western and north-western French Alps, parts on the southern border of the Italian Alps, on the northern border of the Swiss Alps, the northern part of the German Alps, parts of eastern and south-eastern Austrian Alps and on the south-eastern border of the Slovenian Alps (Signer 2010). Higher suitability was predicted for the eastern Alps than for the western and central parts (Signer 2010). Moreover, in the Italian and Austrian Alps larger areas of highly suitable habitat were predicted.

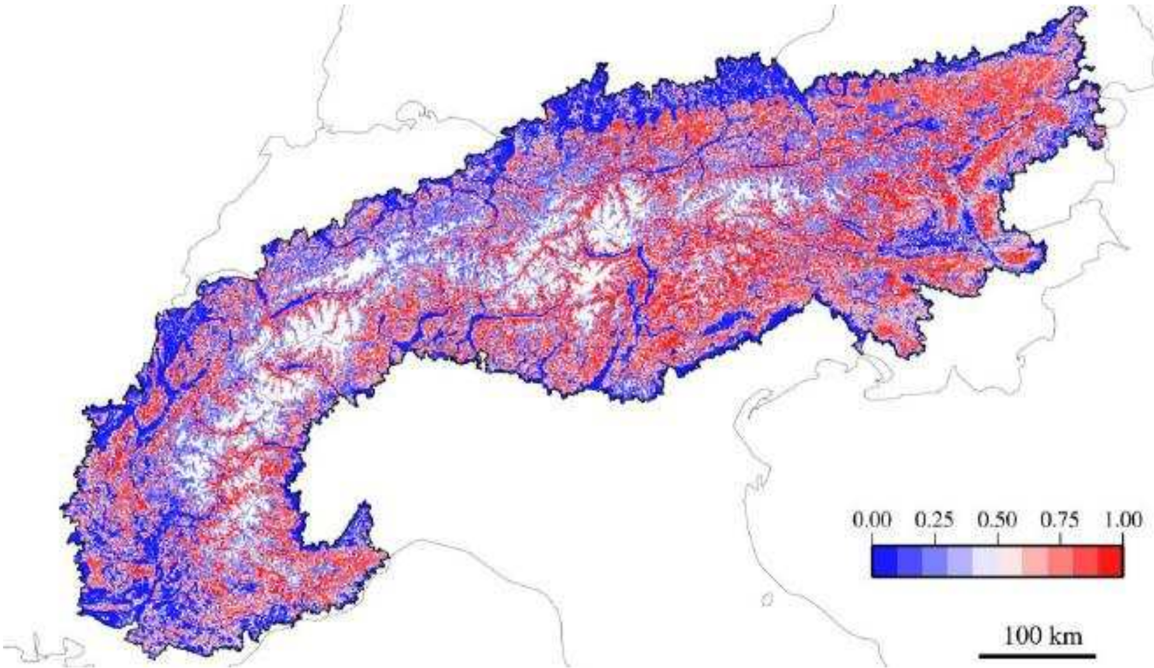


Fig. 7.6. Potential habitat suitability map for lynx in the Alps (Signer 2010).

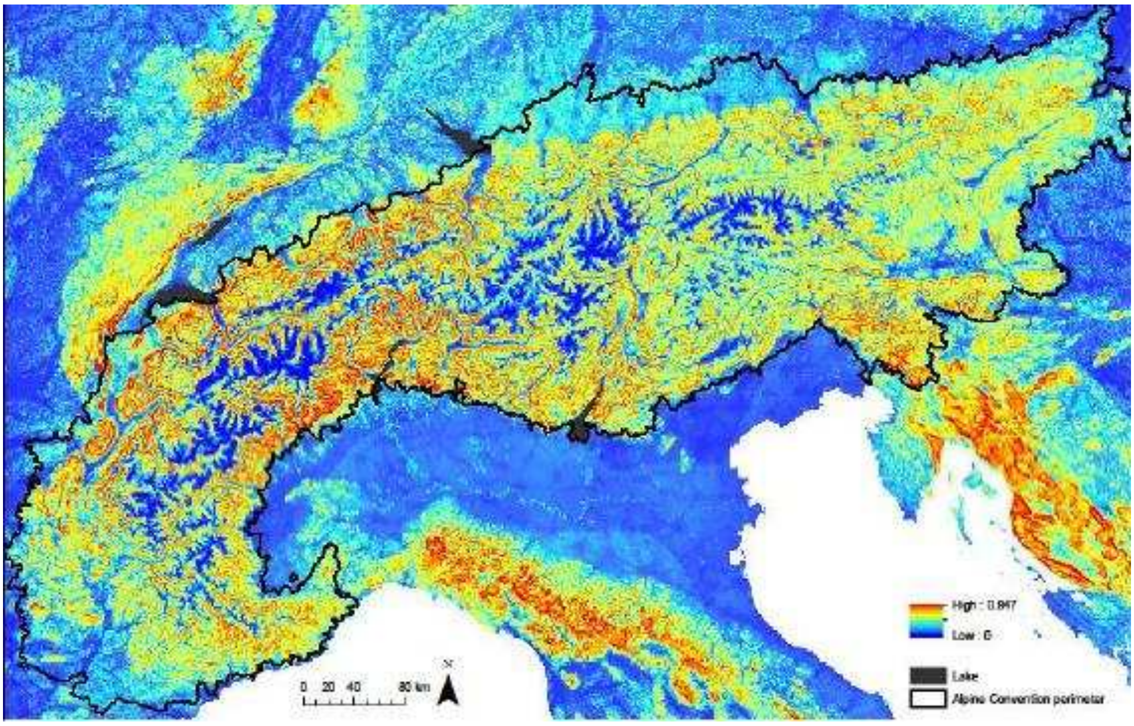


Fig. 7.7. Lynx habitat suitability map based on MaxEnt. Red = highly suitable habitat, blue = low suitable habitat (Becker 2013).

The MaxEnt habitat model of Becker (2013; Fig. 7.7), based on GPS and VHF telemetry data from Switzerland and France and GPS data from Austria and Italy predicted approximately 54% (103,622 km^2) of the Alpine Convention area as suitable lynx habitat. Good suitable habitat for lynx was indicated throughout most of the Alpine range and widespread across all Alpine countries, except at very high elevations (Becker 2013). Large areas with low suitability were indicated in the south-western and north-western French Alps, in parts of the central Italian Alps and on the western border of the Italian Alps, in parts of south-western Switzerland, the northern part of the German Alps, the border of the north-eastern Austrian Alps and in parts of the central, eastern and south-eastern Austrian Alps. Most parts of unsuitable habitat overlapped with areas of very high altitudes. More areas were predicted as highly suitable in the western Alps than in the eastern Alps (Becker 2013). Larger areas of highly suitable habitat were predicted in parts of the northern French Alps, the north-western Italian Alps bordering Switzerland, parts of the Swiss Alps and the south-western Slovenian Alps. The decisive environmental variables could not be identified, as there was no consistent pattern (Becker 2013). The highest uncertainty in the predicted map laid in the French Alps, especially in the southern part, and the southern Italian Alps on the border to France (Becker 2013).

France. No lynx habitat suitability model for the French Alps was found. Rolland et al. (2011) applied two different model methods (Mahalanobis distance factor analysis and site occupancy modelling) on lynx presence data (tracks, scats, chance observations, kills, carcasses, hair) from the French Jura Mountains and compared the results. Road and river density and the proportion of forest were predicted to be the main factors influencing lynx occurrence (Rolland et al. 2011). Higher proportion of forest cover led to a higher suitability and high road and river density to a lower habitat suitability (Rolland et al. 2011). The Ecological Niche Factor Analysis based on lynx presence signs (tracks, hairs, scats, chance observations, carcasses and kills) by Basille et al. (2008) for the Vosges Mountains also indicated that lynx avoid highways and intensively used agricultural areas, but they appear quite unconcerned in regard to the distance to artificial areas (Basille et al. 2008). Thus, it seemed that lynx would support high human activity if there were enough forested areas available. Lynx were predicted to be limited to low agricultural use areas with high forest percentage far from highways (Basille et al. 2008).

Italy. No publication of a habitat suitability model for lynx in the Italian Alps was found.

Switzerland. The model of the north-western Swiss Alps built by Zimmermann (2004) predicted suitable lynx habitat distributed over the whole region, however with the valley bottoms showing up as clearly less suitable (Fig. 7.8). Lynx were predicted to avoid intensive agricultural areas and were mainly associated with forest and other wooded areas (Zimmermann 2004).

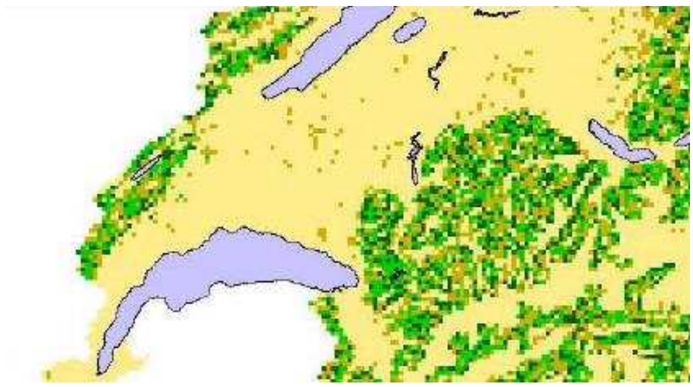


Fig. 7.8. Lynx habitat suitability map for the North-western Swiss Alps (Zimmermann 2004).

The expert models based on Multiple Criteria Decision Making and including an Analytical Hierarchy Process developed by Doswald et al. (2007) for the north-western Alps, based on expert opinions, predicted only parts of the north-western Alps as very highly suitable for lynx, mainly in the southern part of the study area. Habitat found mostly on the edge of the region, was predicted as less suitable. Land cover, forested areas and elevation have been indicated to be important when determining lynx habitat suitability (Doswald et al. 2007). However, the aim of this work was mainly to compare the predictive power of different expert models with a model based on VHF telemetry data. Several expert models successfully predicted habitat suitability for lynx, but they were not finally validated (Doswald et al. 2007). Nevertheless, expert models were found to provide a fast and cheap first approach to predict species distribution if more robust data are not available.

Germany. The habitat model of Schadt (2002) predicted 19% or an area of around 29,119 km² as suitable lynx habitat in the whole of Germany (Fig. 7.9). A home range of a lynx included narrow forest passages or even isolated forest patches if they were less than 1 km apart from each other (Schadt et al. 2002a). Lynx showed a preference for forest and tended to avoid intensively used areas (Schadt 2002). The Alpine area of Germany was predicted as suitable, but not as a source patch, but rather as a target patch for recolonisation; this was however mainly a methodological constraint and a question of the lynx input data (VHF telemetry data from the Jura Mountains; Schadt 2002, Kramer-Schadt et al. 2004).

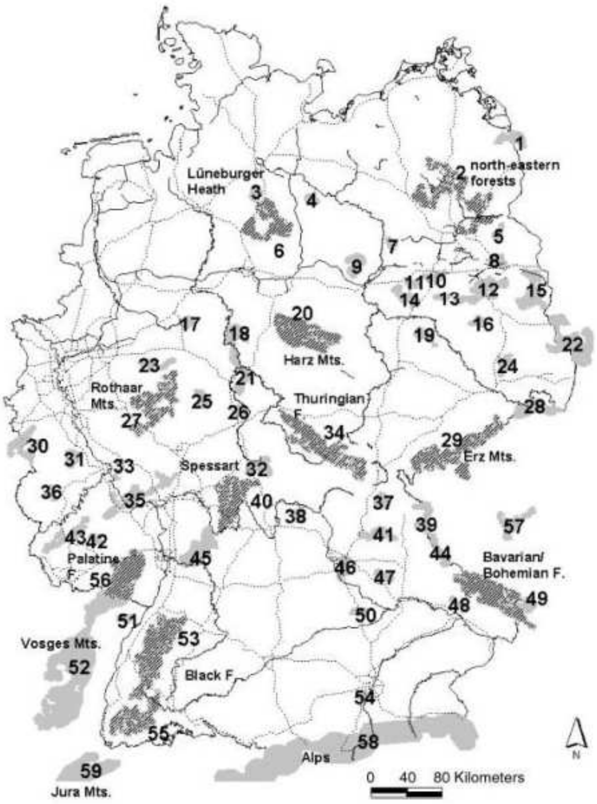


Fig. 7.9 Map of suitable lynx habitat patches in Germany. Dark grey = source patches >1000 km^2, light grey = target patches > 100 km^2, dotted lines = highways, black lines = main rivers (Schadt 2002).

In the habitat model for Baden-Württemberg by Herdtfelder (2012), lynx showed a preference for slopes of up to 50 degrees even if they were located outside of forest areas. Slopes seem to be highly used by lynx as day resting sites. Moreover, lynx preferred forest areas followed by scrubs and open areas (Herdtfelder 2012). Lynx were predicted to avoid the vicinity of roads (Herdtfelder 2012).

Austria. The habitat model by Rüdisser (2001), developed by means of a GIS and based on expert assumptions on the species' ecology, predicted around 59% (11,356 km^2) as suitable lynx habitat in western Austria (Fig. 7.10). The northern and eastern part of the study area (Tyrol, Vorarlberg and western Salzburg) and the south-eastern Tyrol was indicated as more suitable then the rest of the study area. Most of the very low suitable area is indicated in regions with very high altitudes (Rüdisser 2001).

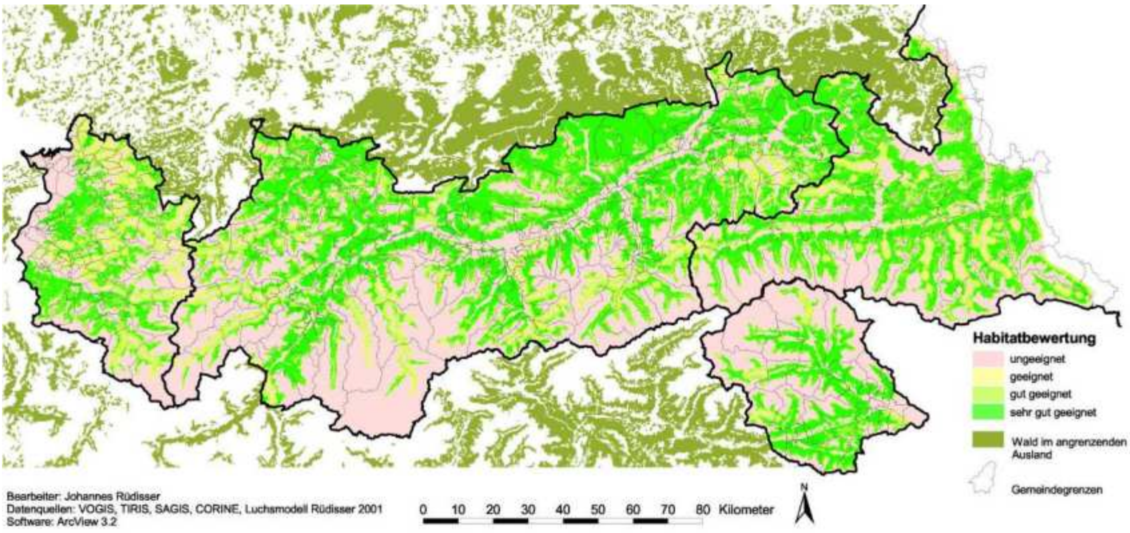


Fig. 7.10. Potential lynx habitat in western Austria (Rüdisser 2001).

Slovenia. No habitat suitability model focussing on the Slovenian Alps was found.

7.2.2. Subpopulations, connectivity and fragmentation

The Alpine range is fragmented mainly by high alpine ridges and densely settled and urbanised valleys, and shows to some degree fragmented forest cover (Zimmermann 2004, Becker 2013; Chapter 5.2.2). The lynx population in the Alps is currently divided into several relatively small genetically isolated subpopulations (Becker 2013; Chapter 4). Connectivity between suitable habitat patches depends on various factors such as the distance between populations, the number of available dispersers in a population and the species' ability to disperse (Zimmermann 2004).

The morphological spatial pattern analysis of Signer (2010) indicated large patches of core areas in the eastern Alps, and the western Alps as more fragmented (Fig. 7.11). Signer (2010) suggests that this is due to the lower altitude of the eastern Alps. Contrarily, the models of Becker (2013) and Zimmermann (2004) predicted larger connected patches of suitable habitat in the central and western parts of the Alps. These differences might be a result of the different data the models are based on (Appendix VI). Both, the habitat suitability map produced by Becker (2013) and Zimmermann (2004) predict a reasonably well connected area of suitable habitat throughout the Alpine range. However, when important barriers (major highways, rivers and high elevation areas), thought to be difficult but not impossible to cross by lynx, are included, the suitable habitat range is fragmented (Fig. 7.12, 7.13). The model by Zimmermann (2004) predicts 37 patches ranging from 50 to 18,711 km^2 in size (patches smaller than 50 km^2 were removed from analysis) with 16 patches over 380 km^2 (Fig. 7.12). The model of Becker (2013) differs slightly; it resulted in a division of 32 patches. Patch sizes ranged from 57–17,378 km^2 with 22 patches >400 km^2, supposed to be large enough to sustain a lynx subpopulation (Fig. 7.13; Becker 2013). Major barriers were defined subjectively based on the experience with radio-collared lynx in Switzerland, thus, patch division and size are only indicative and not definitive (Becker 2013).

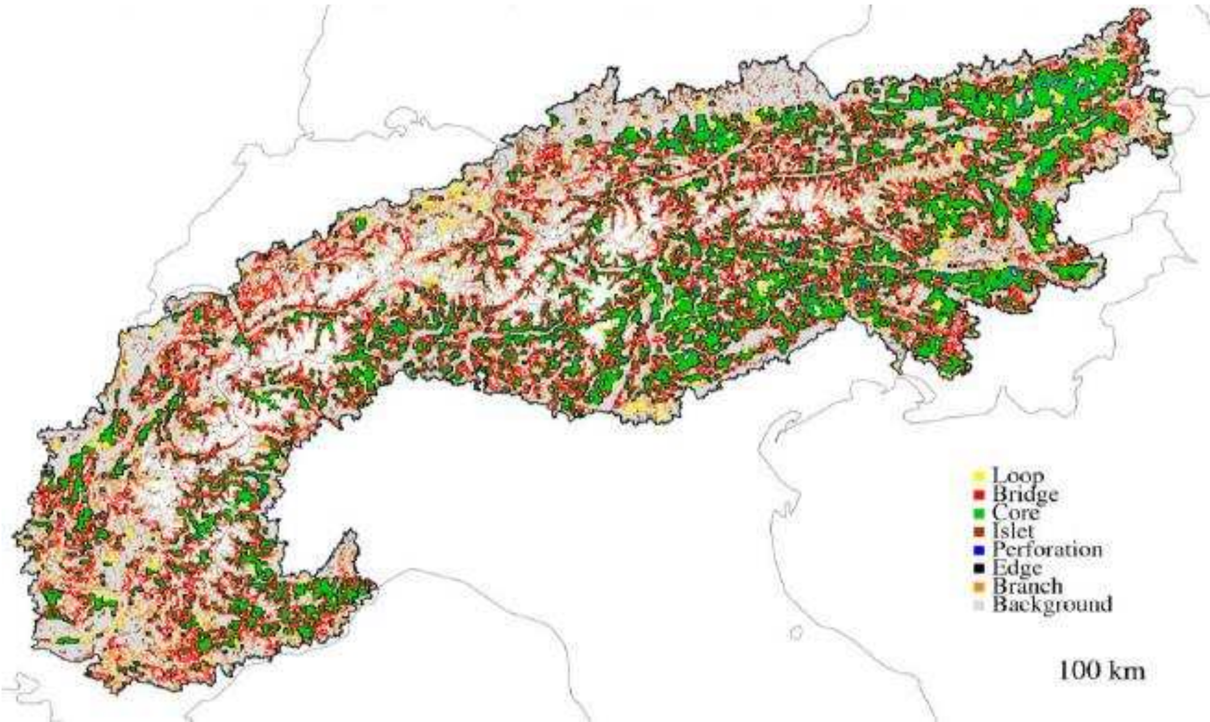


Fig. 7.11. Map of Morphological spatial pattern analysis based on potential lynx habitat suitability (Signer 2010).

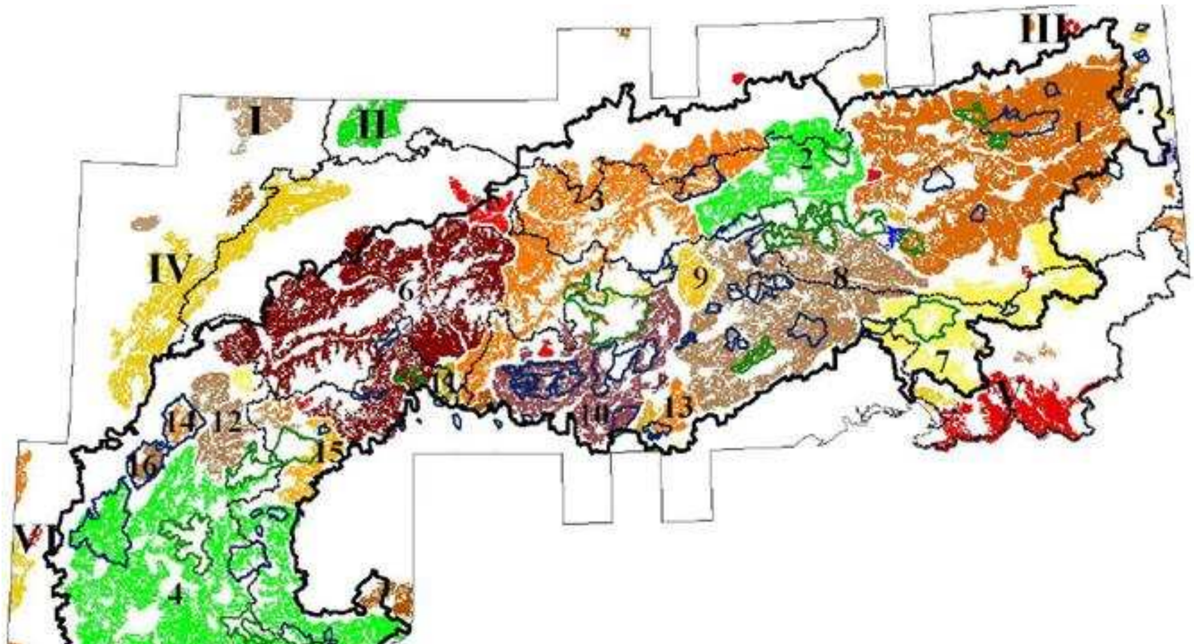


Fig. 7.12. Map of suitable lynx habitat and fragmentation in the Alps and the adjoining regions. Different coloured areas represent distinct patches separated by barriers. Labelled are all patches > 380 km^2 The dark green and blue thick lines delimit the national and regional parks, respectively (Zimmermann 2004).

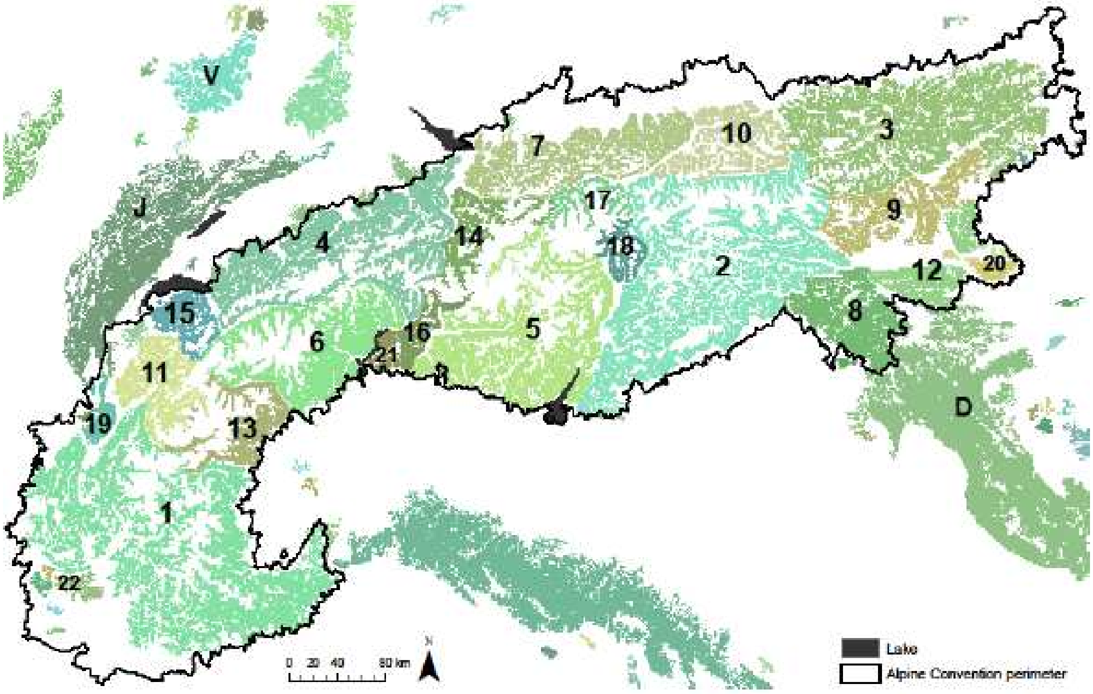


Fig. 7.13. Map of suitable lynx habitat patches after subdivision by major barriers as defined by lynx experts. Each different coloured patch is assumed to represent lynx subpopulations. Patches greater than 400 km^2 are numbered in decreasing order of size. J = Jura, V = Vosges and D = Dinaric Mountains (Becker 2013).

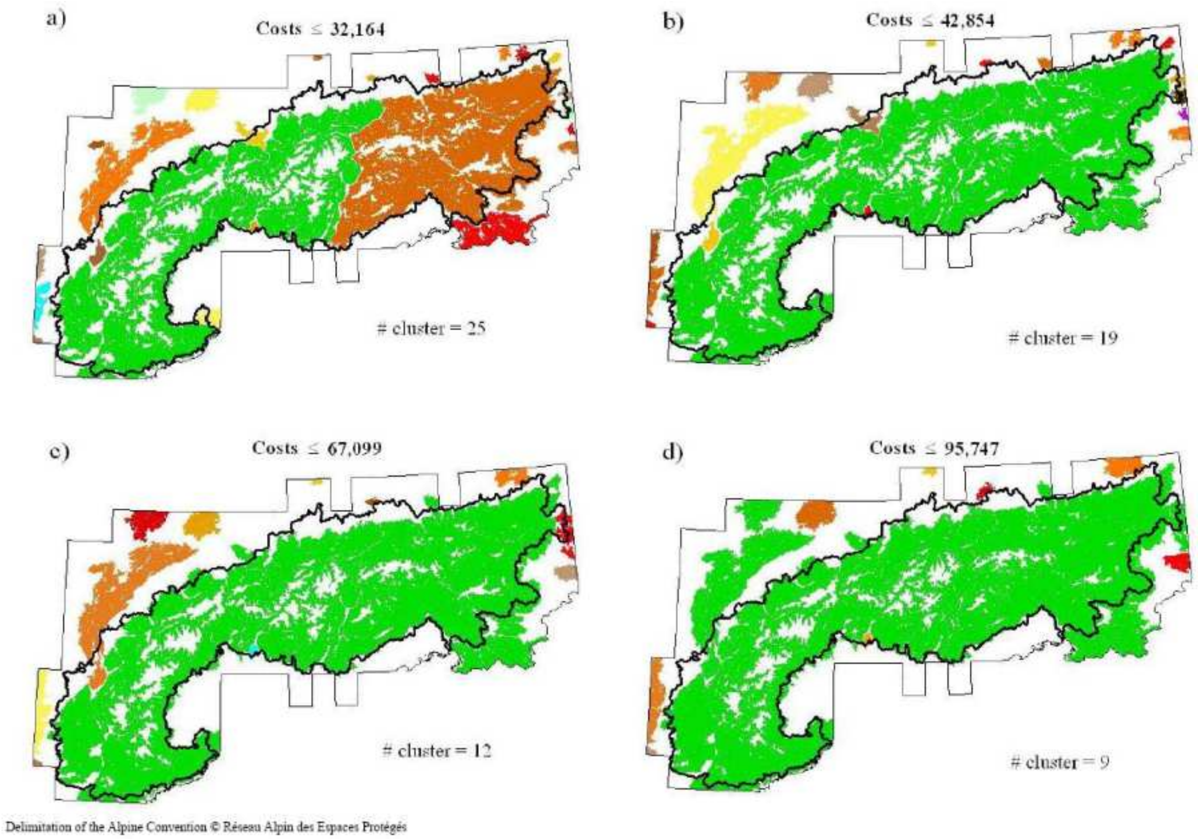


Fig. 7.14. Sensitivity of the landscape to dispersal ability of lynx for the Alps calculated by cost-distance analyses. The patches resulting from the distribution model (Fig. 7.12) are the source patches for the cost-distance function (Zimmermann 2004).

The cost distance analysis of Zimmermann (2004) showed that all defined patches were within the reach of dispersing lynx. Nevertheless, large areas of suitable habitat are only connected by small habitat bands hindering lynx dispersal and operating as bottlenecks (Zimmermann 2004). Depending on the costs to move from one patch to the other, the Alps are predicted to form almost one or two large distinct clusters separated by the Brenner Valley with four smaller, still isolated patches (Fig. 7.14; Zimmermann 2004).

Both, habitat quality and barriers can have an effect on lynx dispersal distances and directions (Zimmermann 2004). Lynx are conservative dispersers and tend to settle in contact with conspecifics (Chapter 7.1; Zimmermann 2004). Barriers such as high mountain peaks, glaciers, highways, large rivers, lakes and settlements are constraints to the expansion of the population (Rüdisser 2001, Schadt 2002, Zimmermann 2004, Becker 2013). The analysis by Signer (2010) revealed a high impact of highways on the lynx distribution in the Alps, but little negative influence of settlements. It was concluded that highways are a major barrier for lynx migration. However, there was no distinction between fenced or unfenced highways and the resistance value was chosen arbitrarily (Signer 2010). Roads are also considered a risk factor by Zimmermann (2004) and thought to be important barriers to lynx dispersal (Zimmermann et al. 2007).

In the Swiss Alps, valleys containing major rivers and many agricultural areas, settlements, roads and railways, as well as the high alpine zones were considered main barriers for lynx expansion by Breitenmoser & Breitenmoser-Würsten (2008). Lynx can physically overcome all kinds of obstacles and can move through unsuitable habitat, but the ability (or will) to cross barriers differed highly between individuals (Zimmermann 2004). Subadult lynx – who are the typical dispersers – apparently were turned back at barriers or unsuitable habitat strips that were traversed by adult lynx (Zimmermann 2004, Breitenmoser & Breitenmoser-Würsten 2008). One male lynx from the north-western Alps crossed a fenced highway several times, once taking an underpass (Zimmermann 2004). Another lynx was recorded to have traversed the Aare valley more than once. This valley includes a high proportion of open habitat, a railway track, a 30 m wide river and a fenced highway (Zimmermann 2004). Nevertheless, in the north-western Alps for example, the dispersal of several lynx was recorded to have been stopped by highways, often leading to “circular dispersal” (Fig. 7.15; Zimmermann 2004, Zimmermann et al. 2007). As individual lynx home ranges never encompass main barriers, they separate individuals and impede the regular contact to neighbours (Breitenmoser & Breitenmoser-Würsten 2008).

The predicted lynx habitat in western Austria is often divided by a broad line of obstacles such as roads, urban areas, rivers or intensive agricultural areas (Rüdisser 2001). The recolonisation of western Austria by lynx from north-eastern Switzerland is hampered by the Rhine Valley with major traffic lines and many settlements (Rüdisser 2009). The Inntal between Telfs and Kufstein with its highway, railway track and urban areas could also be difficult to pass. In the east, the Tauern highway is an important barrier (Rüdisser 2009). These barriers resulted partly in isolated habitat patches in the habitat model (Rüdisser 2001). The model by Kramer-Schadt et al. (2004) indicated that individual suitable lynx patches in Germany are probably isolated mainly due to the dense road and railway system causing high mortality rates (Schadt 2002, Kramer-Schadt et al. 2004).

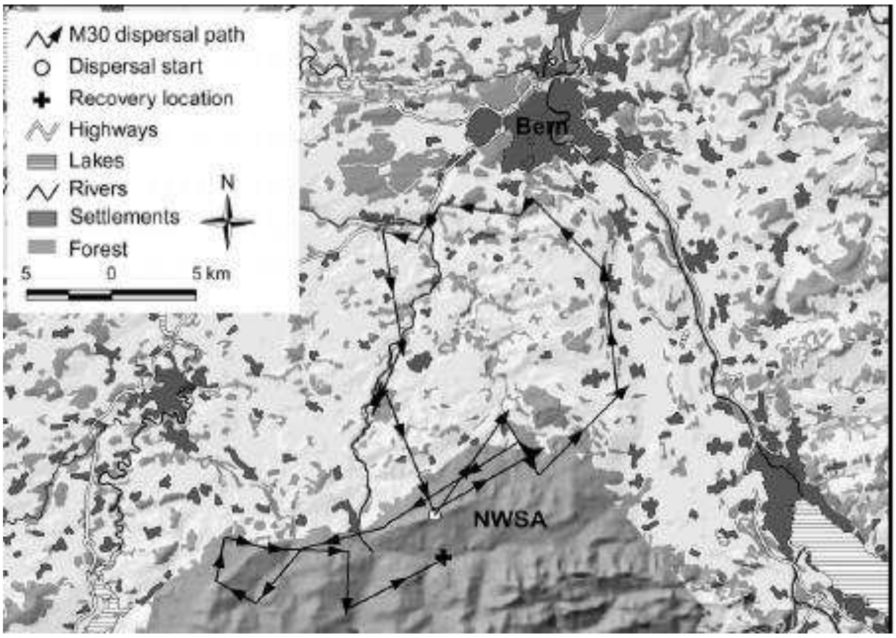


Fig. 7.15. Map of "circular dispersal" route of lynx M30. M30 traversed a sparsely wooded part of the Swiss Plateau and moved north to the vicinity of Bern, where he turned west, following a fenced highway, and spent a week close to the highway and the railway. He finally returned to the north-western Swiss Alps close to where he started his dispersal (Zimmermann et al. 2007).

The Alpine range is, in theory and according to the models, connected to the Dinaric Mountains, the Jura Mountains and to a lower extent the Bohemian-Bavarian lynx populations (Zimmermann 2004). The expansion of the Dinaric lynx population into the Slovenian Alps may have slowed down by urban areas, open habitats and the transport network (Potočnik et al. 2009). Two corridors (Fig. 7.16 corridors E & F) connect the Alps and the Chartreuse, a 688 km^2 mountain which is relatively isolated from the rest of the French Alps (Zimmermann & Breitenmoser 2007), but close to the Jura Mountains. The French Alps are furthermore connected to the Jura Mountains through a corridor via the Salève Mountain (wooded mountain south of Geneva; Fig. 7.16 corridors C1 & C2; Zimmermann 2004). Indeed, a subadult male lynx with a GPS collar used in 2013 exactly this corridor moving from the Swiss Jura Mountains to the French Alps.

7.2.3. Expected abundance of lynx in the Alps

Alpine range

According to estimations based on a MaxEnt habitat model and assuming a lynx density of 1 to 3 independent individuals (resident adults and dispersing subadults; see also Appendix III) per 100 km^2 of suitable habitat, it was estimated that the suitable Alpine area (103,622 km^2) could support approximately 1,035–3,107 lynx (Becker 2013). Zimmermann (2004) applied a lynx density for males of 0.37 – 0.69 and for females of 0.65 – 1.25 individuals (resident lynx) per 100 km^2 of suitable habitat resulting in a slightly lower estimation of lynx for the suitable Alpine area (91,579 km^2) with 961–1,827 individuals (Table 7.1).

Austria. Rüdisser (2001) estimated a potential population of 101–247 lynx (including resident and subadult lynx) for western Austria, based on his habitat model. He used an assumed density of 0.9–2.2 individuals per 100 km^2 of suitable habitat.

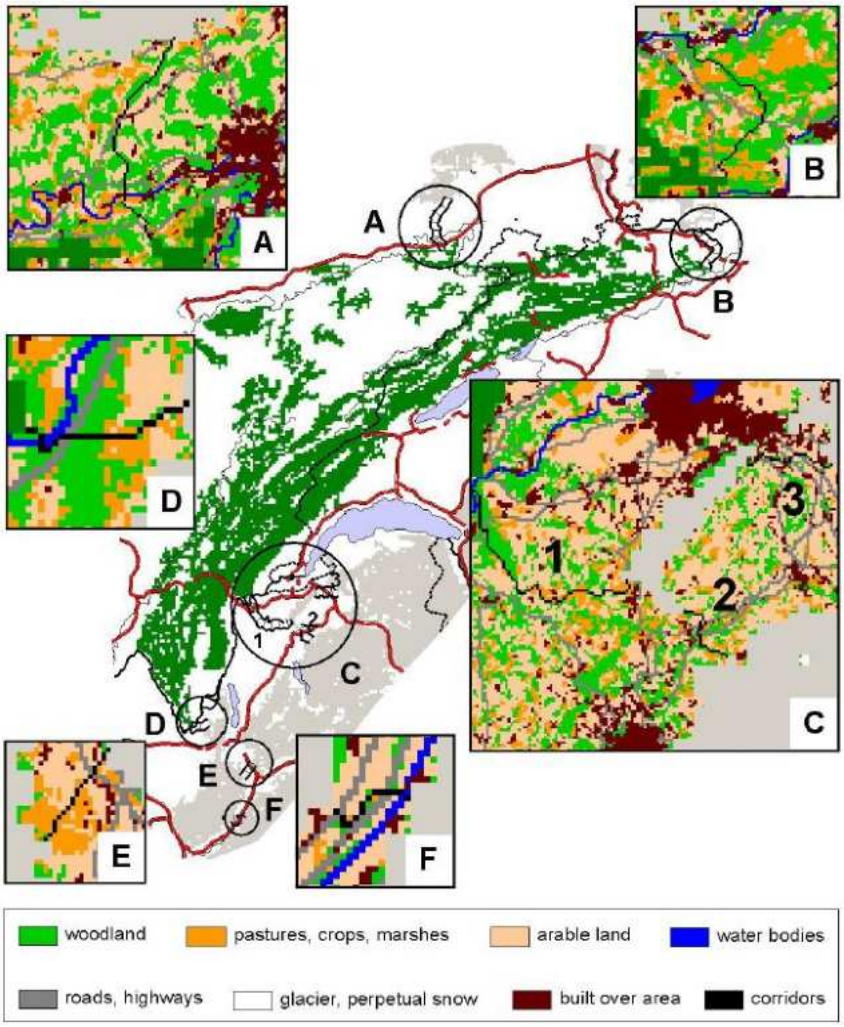


Fig. 7.16. Map of potential corridors between the Jura Mountains and the adjoining areas Vosges Mountains (A), Black Forest (B) and French Alps (C-F). Continuous areas ≥50 km^2 with habitat probability greater than 0.35 are shown in dark green for the Jura Mountains and light grey for the adjacent areas (1x1-km grid) (Zimmermann 2004).

Table 7.1. Estimated number of lynx in the Alpine range and parts of the Alps based on different habitat models.

Author	Density (Ind./100 km^2)	Area of suitable habitat (km^2)	Estimated number of lynx	Remarks
Alpine range				
	1.0		1,035	
Becker (2013)	2.0	103,622	2,075	Independent lynx
	3.0		3,107	
Zimmermann (2004)	0.65 for females and 0.37 for males	93,579	961 (606 females, 355 males)	Resident lynx (adults)
	1.25 for females and 0.69 for males		1,827 (1,174 females, 653 males)	
Western Austria				
Rüdisser (2001)	0.9	11,356	101	resident & subadult lynx
	2.2		247	

7.2.4. Hypothetical expansion of the Alpine lynx population

After reintroductions in the Swiss Alps in the early 1970s, the lynx population expanded rather fast across the north-western Alps until about 1985 and then came to a halt. About 40 years after the first reintroductions, less than 20% of the total suitable habitat in the Alpine region is occupied and the spread of the lynx population appears to have stagnated in spite of the high amount of suitable habitat still available (Molinari-Jobin et al. 2010). Only following translocations and releases to north-eastern Switzerland and the Austrian Kalkalpen, the distribution of lynx expanded slightly (Zimmermann 2004, Becker 2013; Chapter 3.2.2).

The lynx populations in the Alpine range are relatively isolated from each other and only very few migration events between populations occur (Becker 2013). The social structure of the lynx, their need for contact with conspecifics to establish a home range, their dispersal behaviour and the supposed low migration rate between subpopulations, based on cost distance analysis and experience, is thought to be too low to allow the foundation of a new population in a not yet occupied area (Zimmermann 2003, Zimmermann 2004, Becker 2013). Thus, the natural recolonisation of not yet occupied areas by lynx in a fragmented landscape such as the Alps is impeded (Zimmermann 2004, Zimmermann et al. 2007).

A way to overcome such barriers could be translocations and further small-scale reintroduction projects as the one carried out in north-eastern Switzerland or in the Kalkalpen National Park in Austria (Chapter 3.2). Rüdisser (2002) proposed such an approach also for the western Austrian Alps. Reintroduction projects in southern Vorarlberg, eastern Tyrol and western Salzburg would allow establishing further subpopulations and eventually connect the now isolated lynx occurrences in the Alps (Rüdisser 2002).

The lynx subpopulation in north-eastern Switzerland, the new nucleus in the Kalkalpen, the population in the south-eastern Alps and the Bohemian-Bavarian Forest occurrences are believed to be potential sources for the colonisation of the Bavarian Alps (Molinari-Jobin et al. 2010b). However, only the first one was considered large enough to act as a source (Molinari-Jobin et al. 2010b). The likelihood that a lynx reaches the Bavarian Alps depends mainly on the distribution and amount of available suitable habitat and its fragmentation (Herdtfelder 2012). The dispersal of individuals from the Bohemian Forest to the Bavarian Alps is thought to be highly unlikely due to the lack of suitable corridors. Due to the low probability that even a single lynx (e.g. from the north-eastern Swiss Alps) would reach the Bavarian Alps, a natural recolonisation of this area and the establishment of a viable population was considered relatively unlikely (Molinari-Jobin et al. 2010b).

For a colonisation of the entire Alpine range by lynx and to allow genetic exchange, it is necessary to close the gaps between the nucleus in the eastern Alps (Slovenia and Austria) and the one in the north-western Alps (Kaczensky 1998). The north-western Alps population is by far the largest subpopulation in the Alps. However, as a consequence of the very small number of founder individuals some 40 years ago and the slow growth of the population, the population is now strongly inbred. Evidence of genetic drift and reduced heterozygosity are already clearly visible in the lynx population in the north-western Alps (Chapter 4.2.2).

Based on the empirically observed expansion rate between 1995 and 2007, Molinari-Jobin et al. (2010b) estimated that in 2017, only 28,000 km² of the Alps will be occupied by lynx; still less than 20% of the suitable habitat. Although adult lynx survival seems to be the most important demographic parameter for lynx population dynamics, the removal of threats alone will not help to recover a (sub)population, if the connection to or the status of the source population are not sufficient to allow for enough immigration (Zimmermann 2004, Potočnik et al. 2009). For the long-term survival of the lynx in the Alpine range and the conservation of the species, it is crucial to connect the small and genetically isolated lynx subpopulations in the Alps so that they form part of larger metapopulations allowing the exchange of individuals between neighbouring subpopulations and thus to guarantee genetic viability (Rüdisser 2002, Zimmermann 2004, Becker 2013). Natural dispersal alone, depending highly on habitat connectivity, is probably not sufficient to guarantee the expansion (Herdtfelder 2012, Becker 2013). Kramer-Schadt et al. (2011) analysed the effect of "stepping stones" (local lynx population nuclei) and found that they could significantly enhance the colonisation. They however postulated that stepping stones would need to be big enough to produce new dispersers; otherwise they could even negatively impact the colonisation success by binding animals. This is especially noticeable in areas with low to medium dispersal habitats and in cases of high mortality among dispersers. Translocations of individuals may have to be considered (Zimmermann 2004). Without further reintroductions and translocations to new parts of the Alps, it seemed unlikely that the remnant lynx population would expand over the entire Alpine range (Zimmermann 2004).

7.3. Potential distribution, abundance and expansion dynamics of the wolf in the Alps

Some habitat suitability models for wolves for the Alps or part of the Alpine range were developed. Most of them are quite different and based on different methodological approaches, which are summarised in Appendix VI. We will here compile and discuss the outcome of the models, but not the validity of the chosen approach.

7.3.1. Habitat suitability and potential distribution

Alpine Range

Four distinct wolf habitat suitability models for the entire Alpine range are so far available. The MaxEnt habitat model by Herrmann (2011) predicts high habitat suitability over large areas of the Alps and high connectivity over most of the range (Fig. 7.17). Peripheral zones showed lower suitability, e.g. areas in the west of the Rhône-Alpes and the Provence-Alpes-Côte d'Azur region in France, a small strip along the foothills of the Italian Alps, areas south of Lake Geneva and of central Switzerland, most of Lower and Upper Austria, parts of eastern Styria, and southern Carinthia. Almost the whole of the German Alps were indicated as low suitable habitat, as well as parts of the northern and north-western Slovenian Alps (Fig. 7.17). Herrmann (2011) applied a suitability threshold of 0.3 and 0.6, respectively, to tell apart "good" from "very good" habitat, revealing that the best habitat for wolves would be located in the southern French Alps, western and eastern Italian Alps, and the south-eastern Austrian Alps (Fig. 7.18).

Elevation contributed most to the model (Herrmann 2011), whereas forest distribution showed only a low contribution to the model, which was however interpreted as a consequence of its general high abundance over the whole study area (Herrmann 2011). Herrmann (2011) estimated an area of 92,870 km^2 as suitable habitat for wolves, or 49% of the Alps.

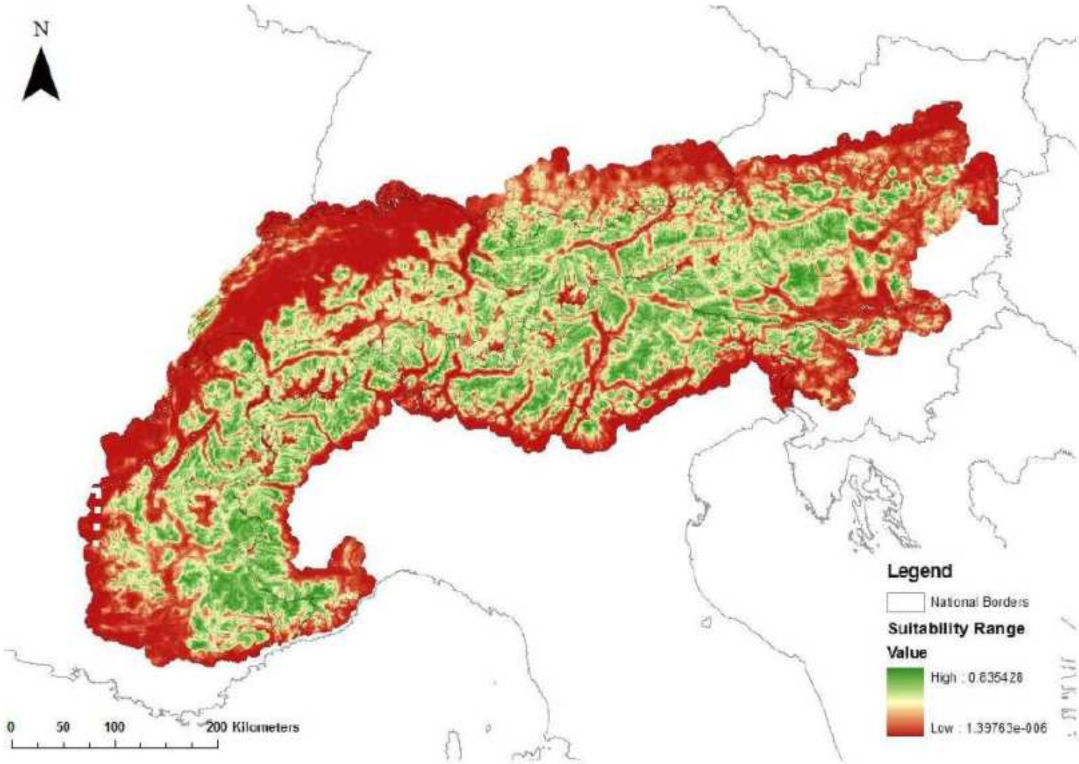


Fig. 7.17. MaxEnt wolf habitat model of Herrmann (2011) based on wolf observations from breeding packs in the French Alps. Suitable habitat = green, unsuitable habitat = red.

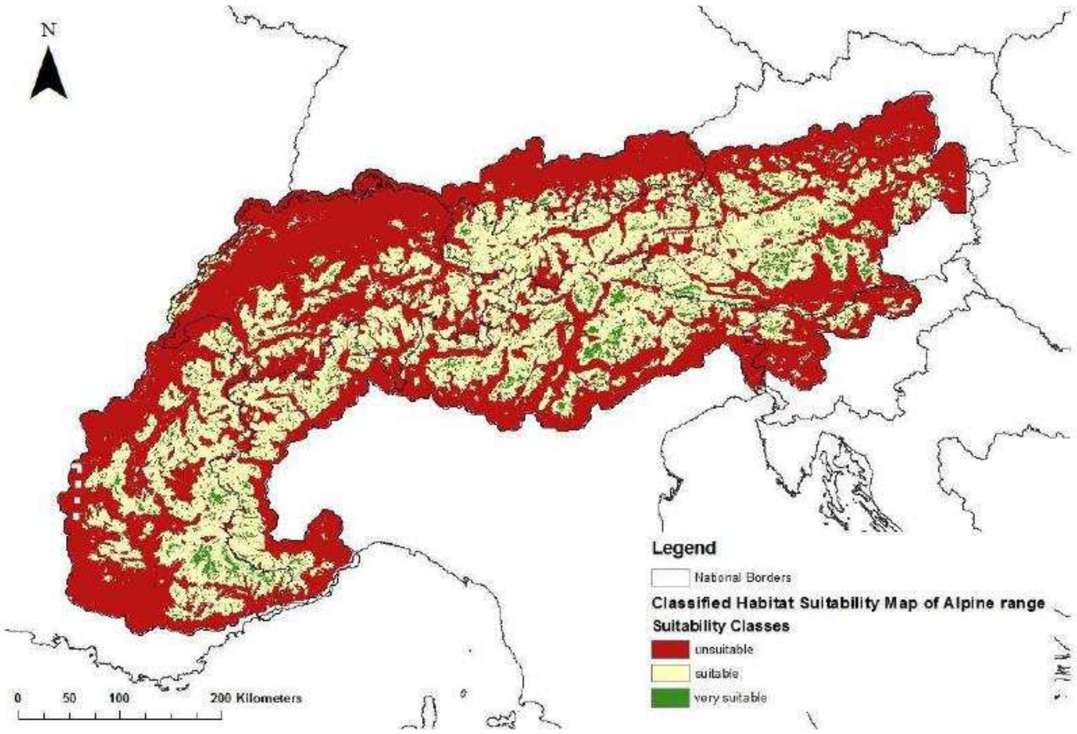


Fig. 7.18. Classified habitat suitability model for the wolf in the Alpine range by Herrmann (2011). The fixed sensitivity approach was used, applying a 5% omission rate (95% sensitivity. i.e. the threshold was chosen in a way that 95% of all presence localities were classified as located in suitable area and 5% were dropped) corresponding to a suitability threshold of 0.3. Additionally, a threshold of 0.6 was used to distinguish “very good” from “good” habitat. Red = unsuitable, yellow = suitable, green = very suitable.

Marucco (2011) developed a wolf habitat suitability model by extending the unconditional multi-season occupancy model of Marucco (2009) for the Italian Alps to the whole Alpine Arc. This model predicted high suitable areas for wolves distributed over the whole Alpine area. Higher suitability is indicated in the eastern and north-eastern Alps than in the western and central-western Alps (Fig. 7.19; Marucco 2011).

Marucco (2011) adapted the spatially explicit, individual-based model (SE-IBM) developed by Marucco & McIntire (2010) and applied it to the entire Alpine range. In comparison to the multi-seasonal occupancy model, the SE-IBM includes the needs of wolf packs and the characteristics of wolf territories to predict habitat suitability of packs (Marucco 2011). The wolf pack habitat suitability map produced by this SE-IBM (Fig. 7.20) looks very similar to the one based on the multi-season occupancy model (Fig. 7.19). However, the SE-IBM predicted larger areas with low suitability, although in the same regions as the multi-season occupancy model. Larger patches with very low suitability were located in the higher elevation areas of France, Switzerland and Austria, north-western and south-western French Alps, in the western Italian Alps, along the southern rim of the Italian Alps, in the northern part of the German Alps and in the south-east of Austria.

The predicted higher suitability of the eastern and north-eastern Alps and lower suitability in the western and central-western Alps was even more pronounced in the SE-IBM (Marucco 2011). These habitat suitability differences are possibly connected to red deer abundance, which is higher in the eastern Alps (Chapter 5.3.1) and to the high elevation in the western Alps. Other prey than red deer was not considered in the models of Marucco (2011). Forest cover and prey presence were predicted to have a positive and human infrastructure (road density and human settlements) and rocky areas a negative effect on wolf presence (Marucco 2011).

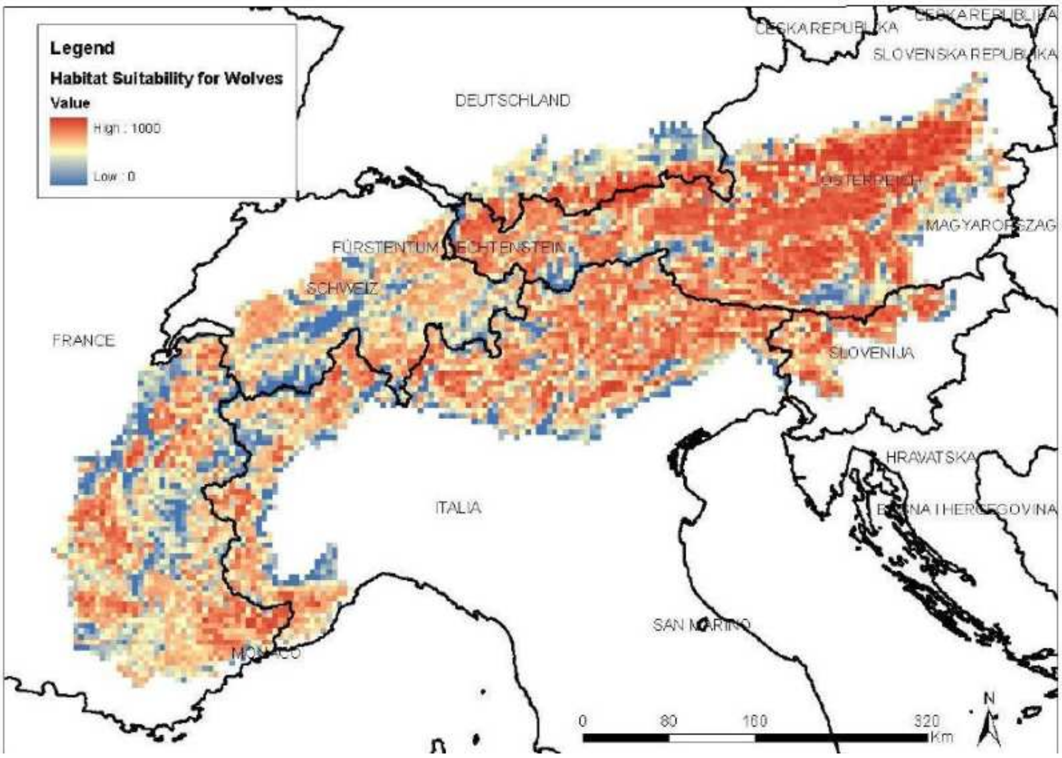


Fig. 7.19. Wolf habitat suitability map based on the multi-season occupancy model (Marucco 2011). Blue = low suitability, red = high suitability.

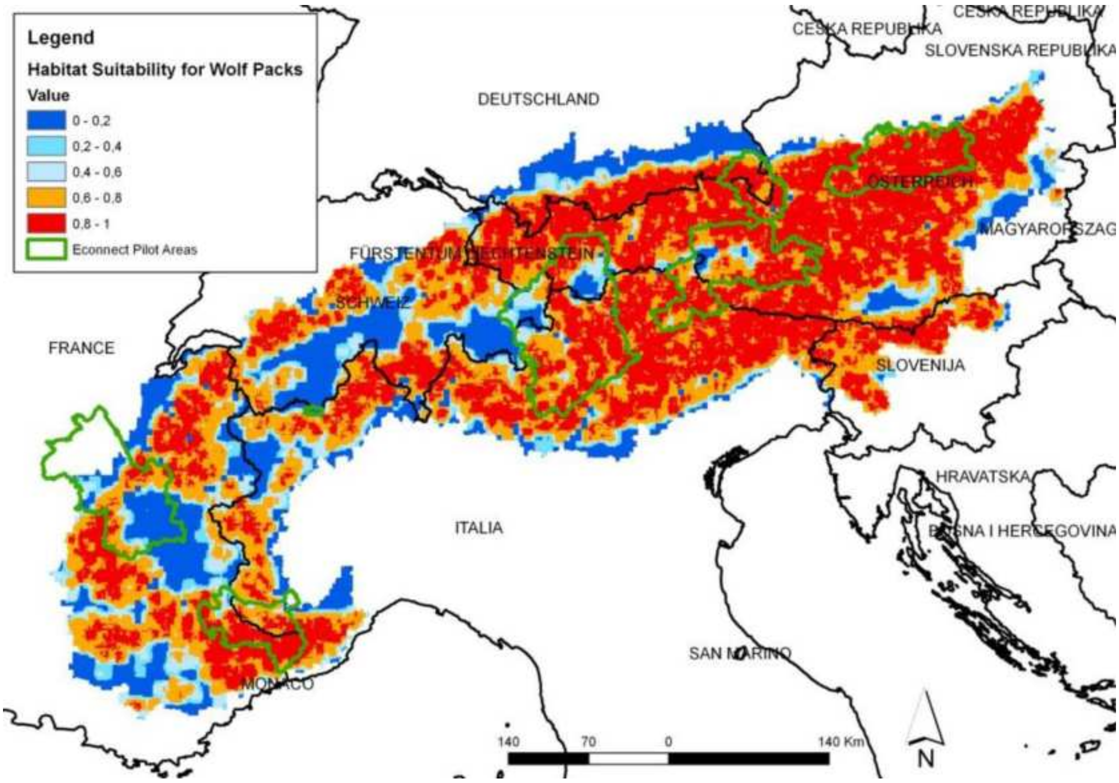


Fig. 7.20. Wolf pack habitat suitability map based on the spatially explicit individual-based model (SE-IBM) developed by Marucco (2011). Blue = low suitability, red = high suitability.

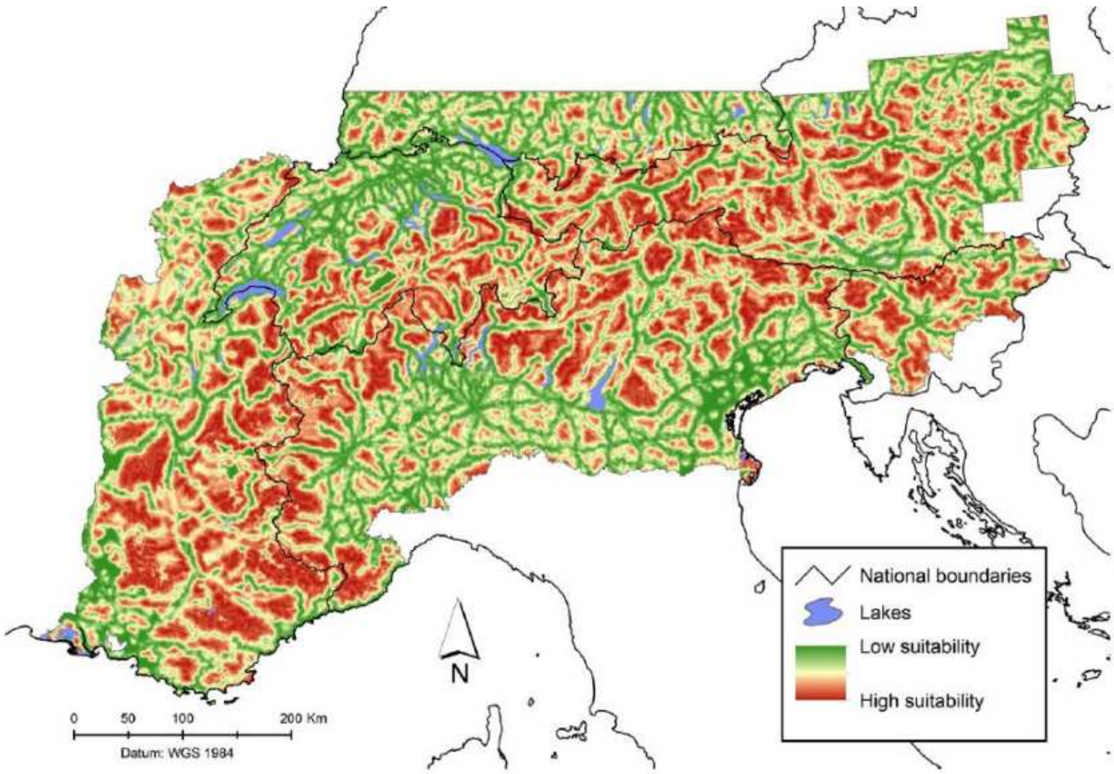


Fig. 7.21. Potential distribution map of the wolf based on the presence-only model developed by Falcucci et al. (2013). Green = low suitability, red = high suitability.

The presence-only partitioned Mahalanobis distance model of Falcucci et al. (2013) predicted almost the whole Alpine range as highly suitable for wolves (Fig. 7.21). Only main valleys and plains, where human density and activity is highest, were shown as less suitable. Areas with the highest suitability were located further from main roads and railways, at an average elevation of 1,603 m and containing natural vegetation, low human population densities (less than 1.2/km^2) and high prey species richness (Falcucci et al. 2013).

France. No habitat model focussing on the French Alps was available. Herrmann (2011) estimated an area of 18,875 km^2 (46% of the Alpine Convention area in France) as suitable for wolves.

Italy. Massolo & Meriggi (1998) used a stochastic model based on logistic regression to determine suitable wolf habitat in the northern Italian Apennines. The dichotomous logistic regression showed a negative effect of hunters' density on wolf presence and a positive of wild boar abundance and scrub extension (Massolo & Meriggi 1998). The polytomous model suggested an important role of north exposure, hunters' density and arable land to limit wolf presence and south exposure, red deer abundance, forest diversity, forest cover, scrub extension and wild ungulate abundance to have positive impact on wolf occurrence (Massolo & Meriggi 1998). Suitable for wolves were habitats where wild prey is abundant and diversified, human impact is low and forest cover is widespread. Massolo & Meriggi (1998) hypothesized that diverse prey systems may provide constant food supply in winter in the absence of livestock, while summer predation on livestock could be reduced due to abundant wild prey and thus reduce the need to search for alternative prey. Their previous model (one-way ANOVA) predicted similar results: human pressure variables were indicated having a negative effect on wolf presence and prey abundance and forest cover making the presence of wolves more probable (Massolo & Meriggi 1996).

Boitani et al. (1998) applied a multivariate analysis over the entire country to evaluate the reliability of using single environmental variables to predict wolf presence in Italy. Only dumping sites and forest expansion showed a significant correlation with wolf presence, but due to poor data for dumping sites, only forest expansion was suggested to be a useful index for wolf habitat quality. Large parts of the Italian Alpine range (mainly in the eastern part) were predicted to be suitable for wolves (Fig.7.22; Boitani et al. 1998).

The discriminant analysis model developed by Corsi et al. (1999) predicted in the Italian Alps an area of around 2,900 km^2 as core wolf habitat (Fig. 7.23). The majority of these core areas were found in the eastern part. Corsi et al. (1999) suggested that human attitude towards wolves was probably the most important factor influencing wolf distribution and density. "Human attitude" was however not available as a GIS layer and hence substituted by other variables such as land use and road density, thus implying to increase linearly with human density. However, this factor could greatly alter with changing attitudes towards wolves (Fig. 7.23; Corsi et al. 1999).

The multi-season occupancy model of Marucco (2009) predicts a high suitability over the whole Alpine range in Italy. Only a strip along the southern rim of the Alps was indicated to be less suitable, as well as parts around the big lakes region (Fig. 7.24). Wolf occupancy rate was explained by landscape, human disturbance and the presence of ungulates (Marucco 2009). Human impact and rock-area cover had a highly negative effect on wolf occupancy (Marucco 2009). Primarily, wolves appeared to avoid humans and rock areas and to occupy the remaining forested and pasture areas (Marucco 2009). Suitable wolf habitat was characterised by the presence of wild ungulates, especially red deer, high proportion of forested area cover and low human impact (roads and settlements), although the overall weight of the forest-cover variable in the model was low (Marucco 2009).

The model of Herrmann (2011) predicted an area of 28,520 km^2 (55% of 51,995 km^2) in the Italian Alps as suitable.

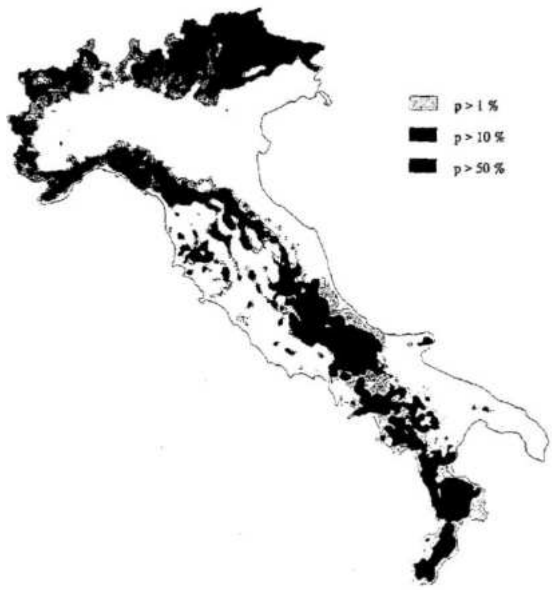


Fig. 7.22. Potential wolf distribution in Italy based on habitat suitability model developed by Boitani et al (1998). Values above 50% indicate suitable areas for wolves.

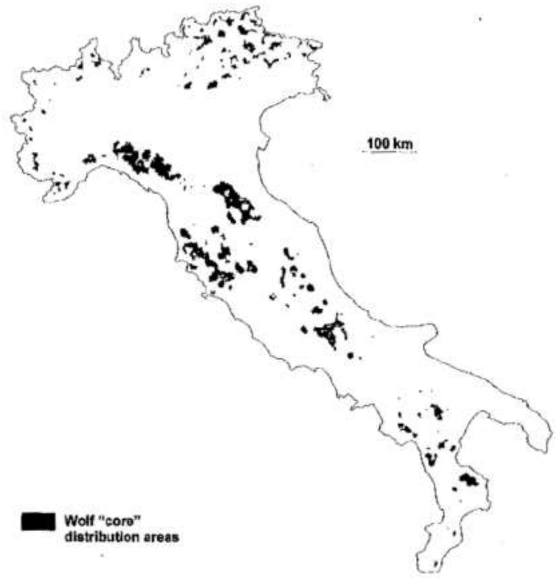


Fig. 7.23. Wolf core area distribution obtained by discriminant analysis model developed by Corsi et al (1999).

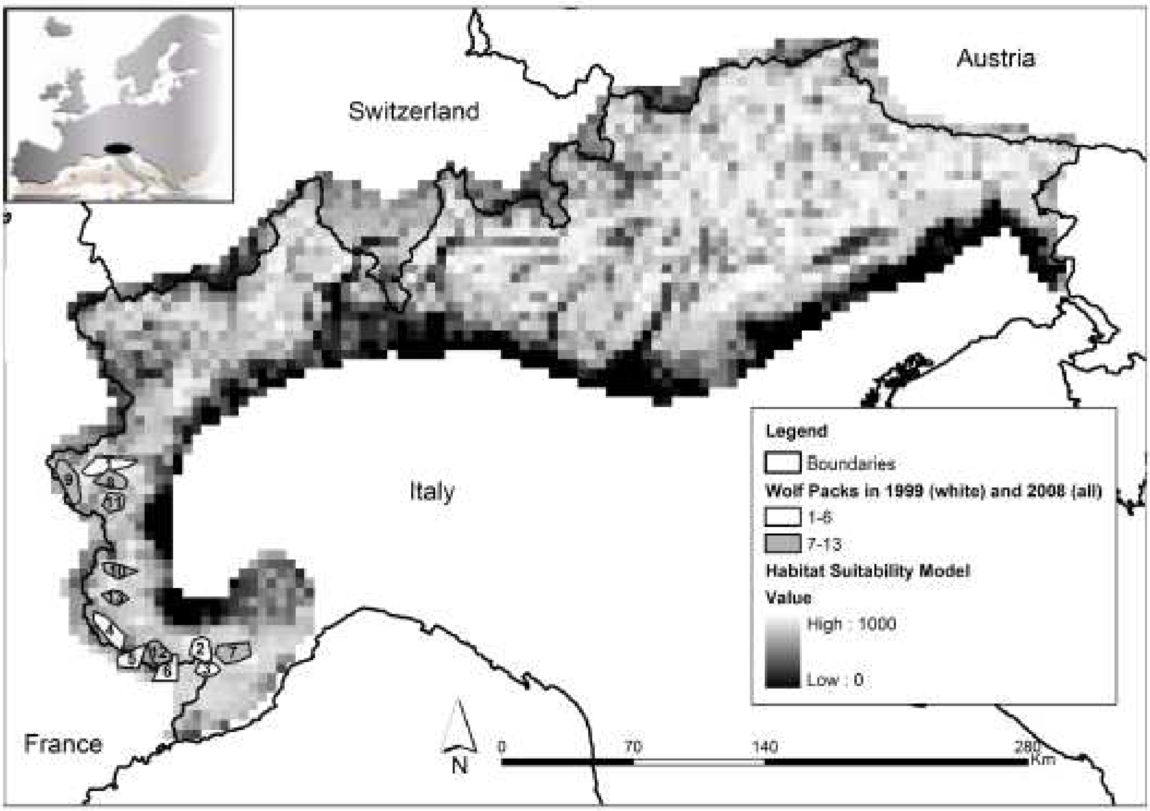


Fig. 7.24. Wolf habitat suitability model in the Italian Alps based on multi-season occupancy model developed by Marucco (2009). The locations of the first six wolf packs in the South-Western Alps and the locations of the 13 wolf packs in the Western Alps are illustrated (Marucco 2009).

Switzerland. Three different habitat models which include different parts of the Swiss Alpine range have been developed. The stochastic logistic regression models by Glenz (1999) and Glenz et al. (2001) analysed the habitat suitability of the Canton of Valais, where the wolf arrived first. It predicted an area of 19% (1,143 km^2) of the Canton as suitable and 606 km^2 as highly suitable habitat for wolves (Glenz et al. 2001). These areas mainly follow the forests along the main and smaller side valleys whereas the bottom of the valleys is indicated as less suitable; areas below 800 – 900 m are predicted as much less suitable for wolves due to high human pressure in low areas (Fig. 7.25; Glenz et al. 2001). Areas above 1800 –2000 m are predicted to be less suitable due to the lack of prey in high elevations. Based on the simple correlation analysis a negative impact of roads on habitat suitability was detected, but the strongest influence on the model was shown by the wild ungulate diversity index. Glenz et al. (2001) concluded that with enough prey available, wolf presence would mainly be limited by human presence.

An area of 260 km^2 was predicted as suitable for wolf reproduction. These areas again stretched along the main and the side valleys (Glenz et al. 1999). These areas seemed to be quite isolated one from another, although there were some connected areas suitable for wolf reproduction in the upper Valais (Fig. 7.26). The results of these two models are more conservative than all other models, but have to be interpreted with caution, as they used a logistic regression approach and did not account for spatio-temporal factors. Moreover, it was based on data from the Apennines (Italy) with distinct environmental conditions from the Valais (Glenz 2001).

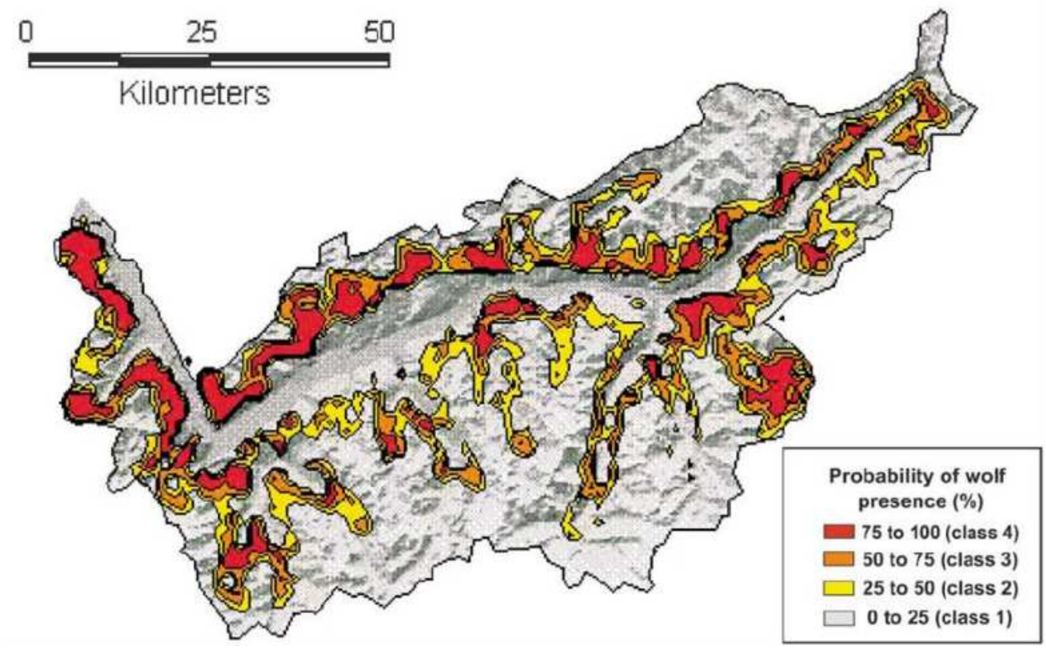


Fig. 7.25. Wolf habitat suitability map of the Canton of Valais, Switzerland created by Glenz et al. (2001).

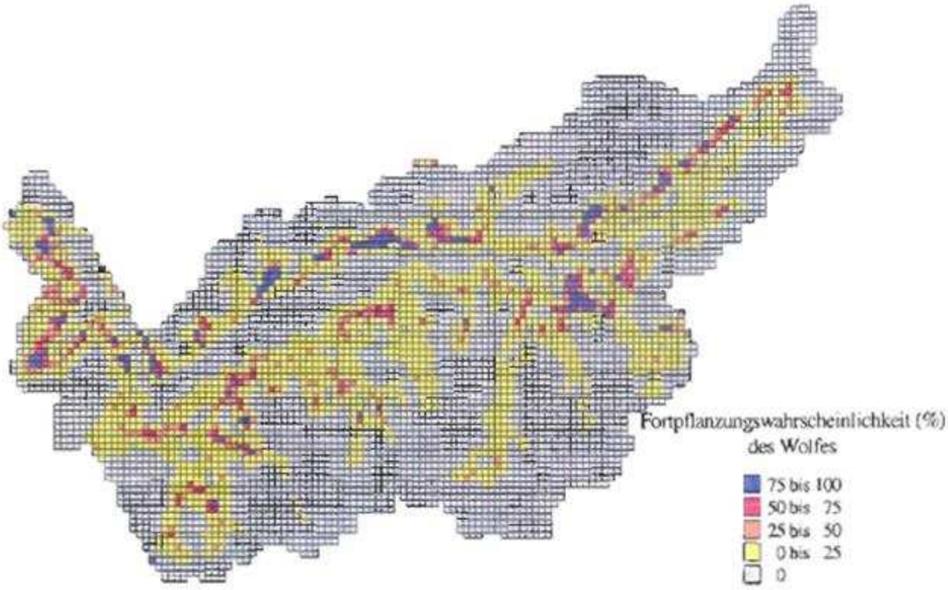


Fig. 7.26. Map of wolf reproduction probability Glenz (1999).

The GIS based model of Landry (1997b) evaluated the habitat suitability of the southern Swiss Alps (Cantons of Valais, Ticino and Grisons), based on five different variables. It hypothesized that wolves would mostly use areas below 3,000 m in summer and areas below 2,000 m in winter (Landry 1997b). In summer, more than 79% (15,142 km^2) and in winter around 31% (5,942 km^2) of the study area was predicted as suitable wolf habitat (Fig. 7.27a, b). Landry (1997b) concluded that the suitable habitat for wolves was determined by the available habitat in winter. Yet, the model was based on assumptions on the species preferences and not on field data and the results were therefore arbitrary.

Fig. 7.27a. Predicted summer habitat for wolfs in the southern Swiss Alps based on the following assumptions: at least 75% forested area, less than 30 people/km^2, at least 22% area of alpine pastures, not more than 16% agricultural area, and less than or equal to 1% urban zone (Landry 1997b).

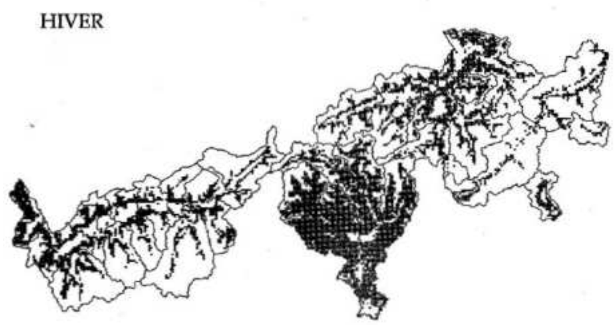


Fig. 7.27b. Predicted winter habitat for wolfs in the southern Swiss Alps based on the following assumptions: at least 75% forested area, less than 30 people/km^2 at least 22% area of alpine pastures, not more than 16% agricultural area, and less or equal 1% urban zone (Landry 1997b).

The MaxEnt model of Herrmann (2011) predicted suitable areas for wolves in the Swiss Alps mainly along the slopes of valleys, hence along the band of forests. High mountainous areas and valleys were less suitable and suitability decreases with increasing human disturbance (urban area density, agricultural land, road density; Fig. 7.28a, b). Depending on the data used (French wolf monitoring data or Swiss chance observations) the predicted suitable habitat differed slightly. These differences are best visible in the Ticino and the Grisons (Herrmann 2011). An area of 12,020 km^2 is predicted to be suitable wolf habitat in entire Switzerland according to the habitat model based on the data from France and around 11,845 km^2 as suitable habitat based on the data from chance observations in Switzerland (Herrmann 2011). Applying a threshold of 0.3 and 0.6 to better distinguish between suitable and unsuitable habitat, the areas of best habitats are restricted to few high patches (Fig. 7.28c, d). Data from France included the habitat use by packs based on systematic monitoring, whereas the chance observations in Switzerland were almost entirely chance observations from solitary male wolves. Applying the more "wolf-like" data from the French monitoring predicted more suitable habitat in the Swiss Alps than applying the local, but observer-biased Swiss chance observations.

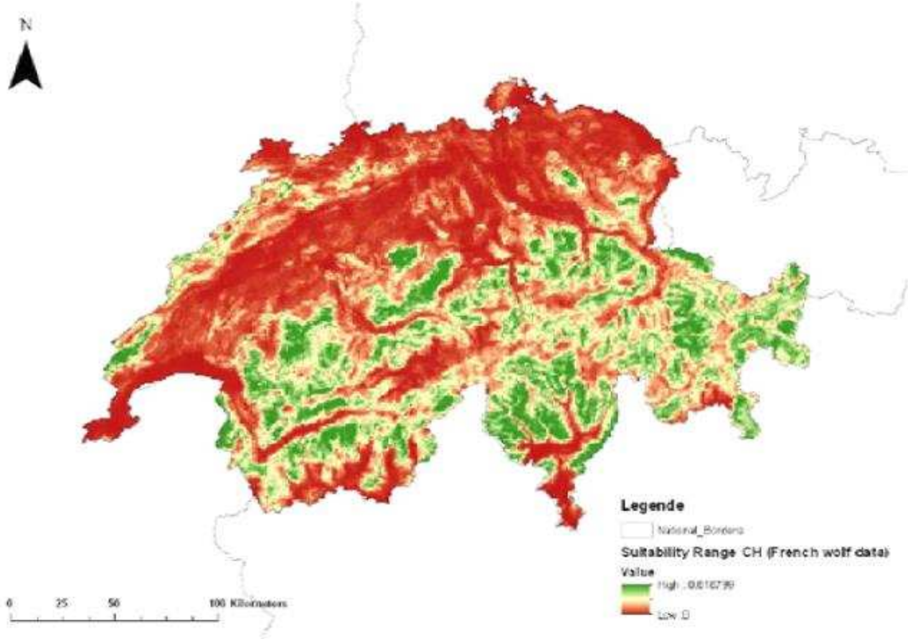


Fig. 7.28a. Continuous habitat suitability map of Switzerland based on data from the French wolf monitoring study (Herrmann 2011). Red= low suitability, green = high suitability.

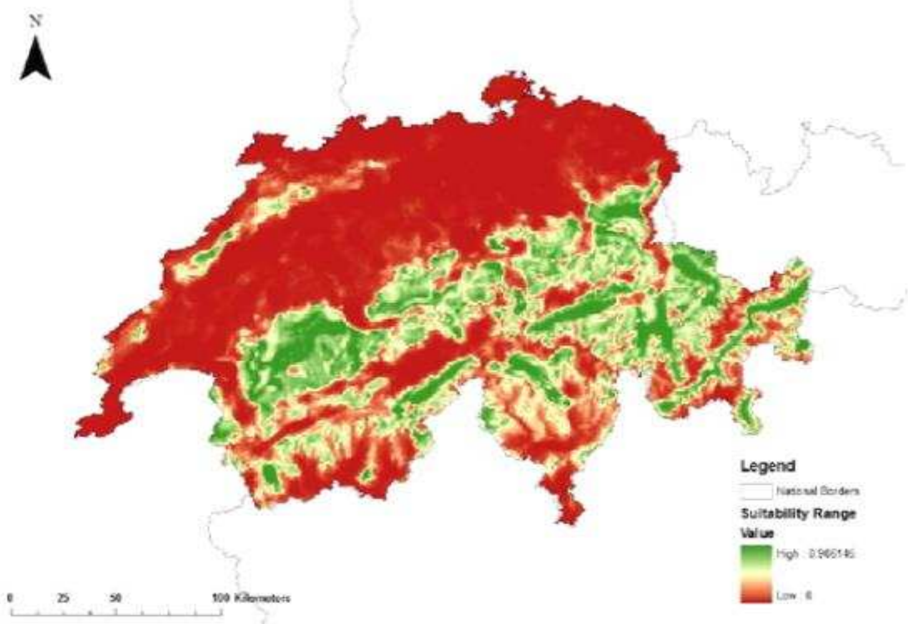


Fig. 7.28b. Continuous habitat suitability map of Switzerland based on data from chance observations from Switzerland (Herrmann 2011). Red= low suitability, green = high suitability.

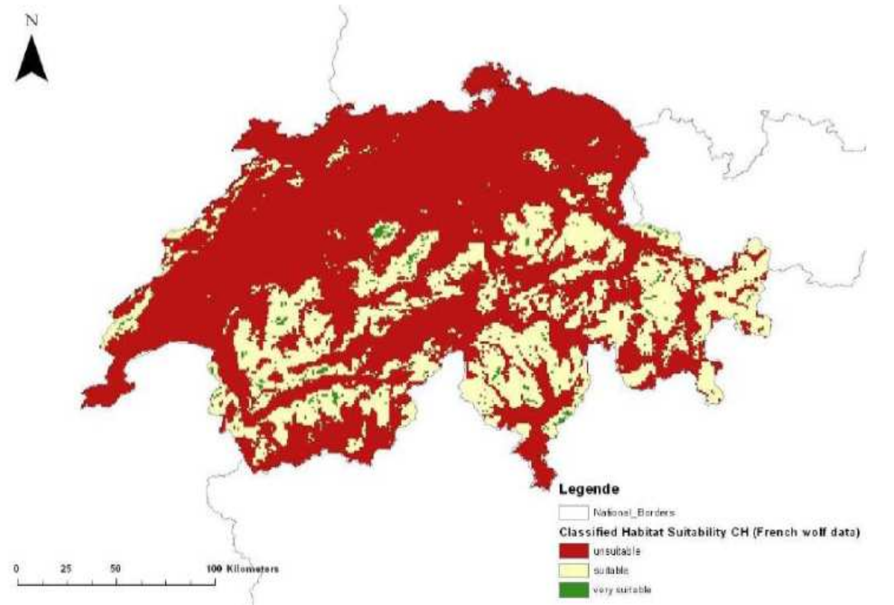


Fig. 7.28c. Continuous habitat suitability map of Switzerland based on data from French wolf monitoring studies, with an arbitrary threshold of 0.3 and 0.6 (Herrmann 2011) to tell good from best habitat. Red = unsuitable, yellow = suitable, green = very suitable.

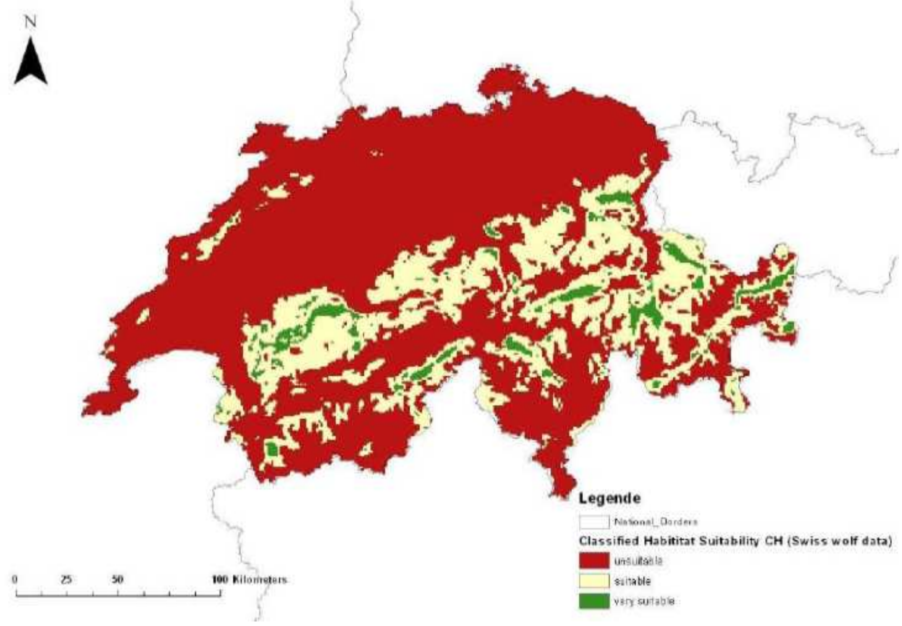


Fig. 7.28d. Continuous habitat suitability map of Switzerland based on data from chance observations in Switzerland with an arbitrary threshold of 0.3 and 0.6 (Herrmann 2011) to tell good from best habitat. Red = unsuitable, yellow = suitable, green = very suitable.

Germany. The spatially-explicit predictive rule-based models by Fechter & Storch (2014), which were developed for the whole of Germany, predicted all the Bavarian Alps as highly suitable wolf habitat (Fig. 7.29). The model of Fechter & Storch (2014) indicated a wide range of wolf habitat types but revealed a preference for forest areas. Wolf presence was limited to areas with low road densities; wolves were predicted to avoid settlements and areas of human activity, and to establish home ranges in areas with the least human disturbance. However, these temporal, spatio-temporal or behavioural avoidance characteristics may change with habituation. Wolves can adapt to densely human populated areas with high road densities if they are tolerated by humans (Fechter & Storch 2014). The authors concluded that most likely the attitude of humans towards wolves would restrict wolf presence in Germany. The results of this study have to be interpreted with care, as the primary aim of the work was to examine the uncertainty in rule-based habitat models when applied for habitat generalists such as the wolf, and not to predict habitat suitability of the species.

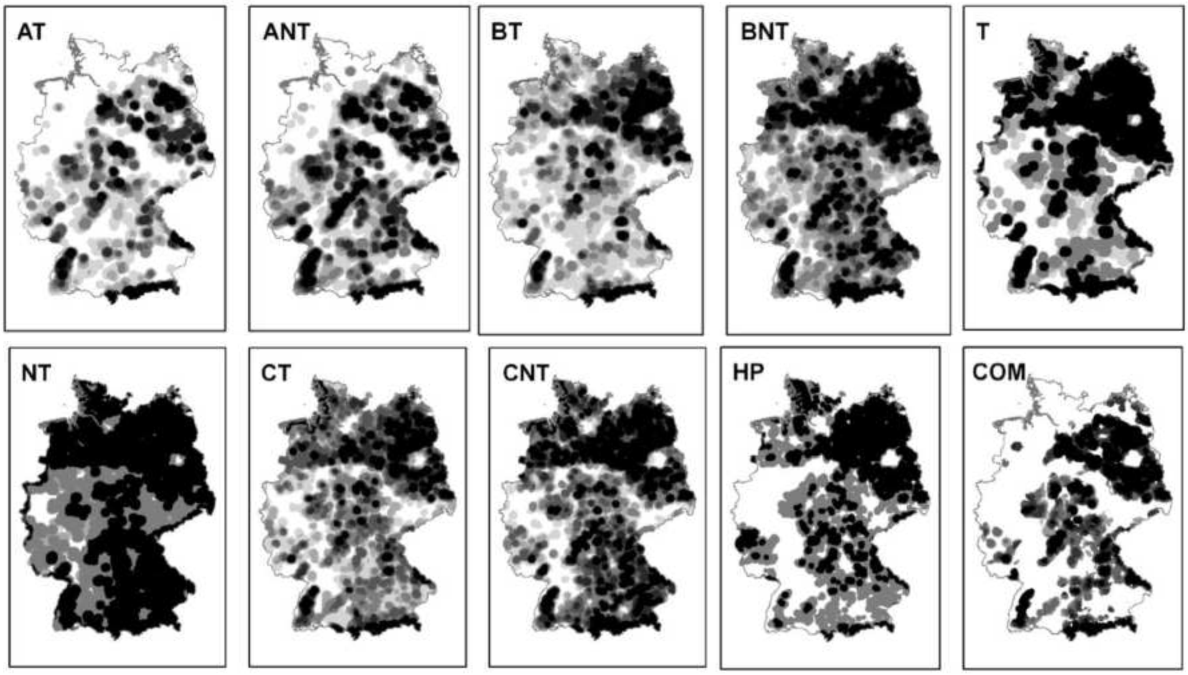


Fig. 7.29. Wolf habitat suitability maps produced by Fechter & Storch (2014). Each map was produced by rule-based models including different environmental parameters.

Herrmann (2011) estimated the area of suitable wolf habitat in the German Alps at 2,610 km^2 (23% of the Bavarian Alpine Convention area).

Austria. There is no habitat suitability model focussing on Austria alone. The study of Herrmann (2011) estimated an area of about 28,880 km^2 (53% of the Alpine Convention area in Austria) as suitable wolf habitat.

Slovenia. No habitat suitability model focussing on the Alpine range in Slovenia was available. Herrmann (2011) estimated an area of 1,900 km^2 (28% of the Alpine Convention area in Slovenia) as suitable wolf habitat.

7.3.2. Subpopulations, connectivity and fragmentation

No subpopulations of wolves were identified in the Alpine range. The wolf population in the Alps was considered to be a distinct population unit (e.g. Linnell et al. 2008) as it was defined to be different in ecological and socio-economic contexts, although it is demographically and genetically connected to the wolf population in the Italian Apennines (Marucco 2009), as well as to the Dinaric and north-eastern European populations. From a conservation management perspective, the Alpine wolf population is also considered special as it ranges over different countries (Marucco 2009). Wolf habitat connectivity and fragmentation is most likely impacted by natural and anthropogenic factors (Marucco 2009; Chapter 7.3.1).

Areas of high human activity do not seem to present a major barrier to wolf dispersal if wolves are tolerated by humans (Valière et al. 2003). Wolves can easily traverse roads and highways, and a single road does not act as a barrier (Ciucci et al. 2009, Marucco 2011). A wolf, for example, which moved from the Apennines (Italy) to the western Alps in France, crossed four fenced four-lane highways and several main railways (Ciucci et al. 2009). This wolf traversed open habitat, agricultural and developed areas but used often underpasses to cross highways (Ciucci et al. 2009). A male wolf dispersing from Slovenia to Italy crossed several natural and anthropogenic barriers such as highways, railways, unsuitable habitats and mountain ridges (WAG 2014). Nevertheless, if road density is high, it can limit pack settlement (Marucco 2011). Wolves get killed quite often in traffic accidents if the wolf pack territory includes many roads (Marucco 2011). Road density implies several other factors such as human presence and effects of roads (and people using the roads) on prey species and wolves. Roads not only pose a risk of mortality but can also lead to reduced habitat quality through fragmentation or by providing easy access to wildlife areas to people (Marucco 2009, Fechter & Storch 2014).

The wolf population in the south-western Alps is connected to its source population in the Apennine through the Ligurian Apennine Mountains, acting as an ecological corridor also important to assure genetic diversity (Marucco 2009, Marucco & McIntire 2010, Marucco 2011). The Alpine wolf population is furthermore connected to the Dinaric population via Slovenia, as demonstrated by the genetic identification of Dinaric wolfs in Austria and Italy (Marucco 2011; see also Fig. 7.4 in Chapter 7.1). Wolves can also immigrate to the Alps from the Carpathian and from the Central European Lowland or north-eastern European wolf populations, respectively, making the future Alpine population a melting pot for nowadays several genetically distinct populations.

Falcucci et al. (2013) predicted a high connectivity of suitable wolf habitat across the whole Alpine range. However, this prediction has to be considered carefully as the model did not account for spatial or temporal aspects or social structures or pack requirements. The morphological spatial pattern analysis of Marucco (2011) based on SE-IBM developed by Marucco & McIntire (2010) taking into account such aspects, classified (using thresholds of 0.5 and 0.8 chosen arbitrarily to detect more important core areas) only around 70% or 25% (48,357 km^2) of the Alpine range as wolf core areas (Fig. 7.30a, b; Marucco 2011). Depending on the selected threshold, the degree of connectivity between wolf core areas varies highly (Marucco 2011). A network analysis with threshold of 0.5 indicates a large connected area over most of the Alpine range. Applying a threshold of 0.8 shows however a different picture, with high fragmentation indicated especially in the western-central Alps, whereas the eastern part of the Alps is still predicted as highly connected (Fig. 7.31a, b; Marucco 2011). The main unit of the analysis for the model of Marucco (2011) were wolf packs, not lone wolves.

According to the cost-distance analysis developed by Marucco (2011), most potential natural and anthropogenic barriers for movements of wolves, were identified in the western-central Alps and in Switzerland (Fig. 7.32). Roads, settlements, high altitude areas, low forest coverage and lakes were considered as potential barriers. The lowest level of connectivity has been identified between source areas in the Pennine and Lepontine Alps, between Italy and Switzerland (Marucco 2011). These areas with high barrier density may slow down the spread of the population in the south-western Alps and may lead to a more erratic expansion in the central and eastern Alps rather than the continuous advance of the “population front”. But it does – as empiric observations over the past 20 years demonstrate – not principally obstruct the dispersal of wolves.

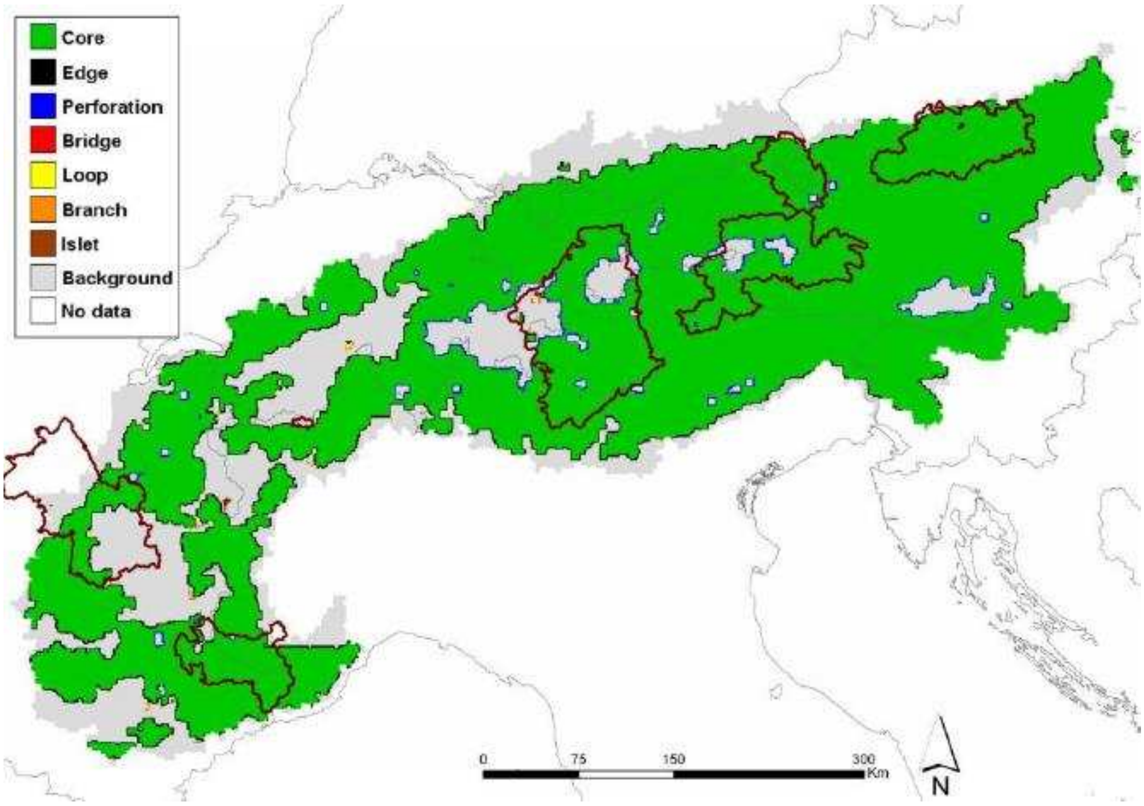


Fig. 7.30a. Morphological spatial pattern analysis based on wolf pack habitat suitability map for the Alpine range. A threshold value of 0.5 was applied. Brown polygons = Econnect pilot regions (Marucco 2011).

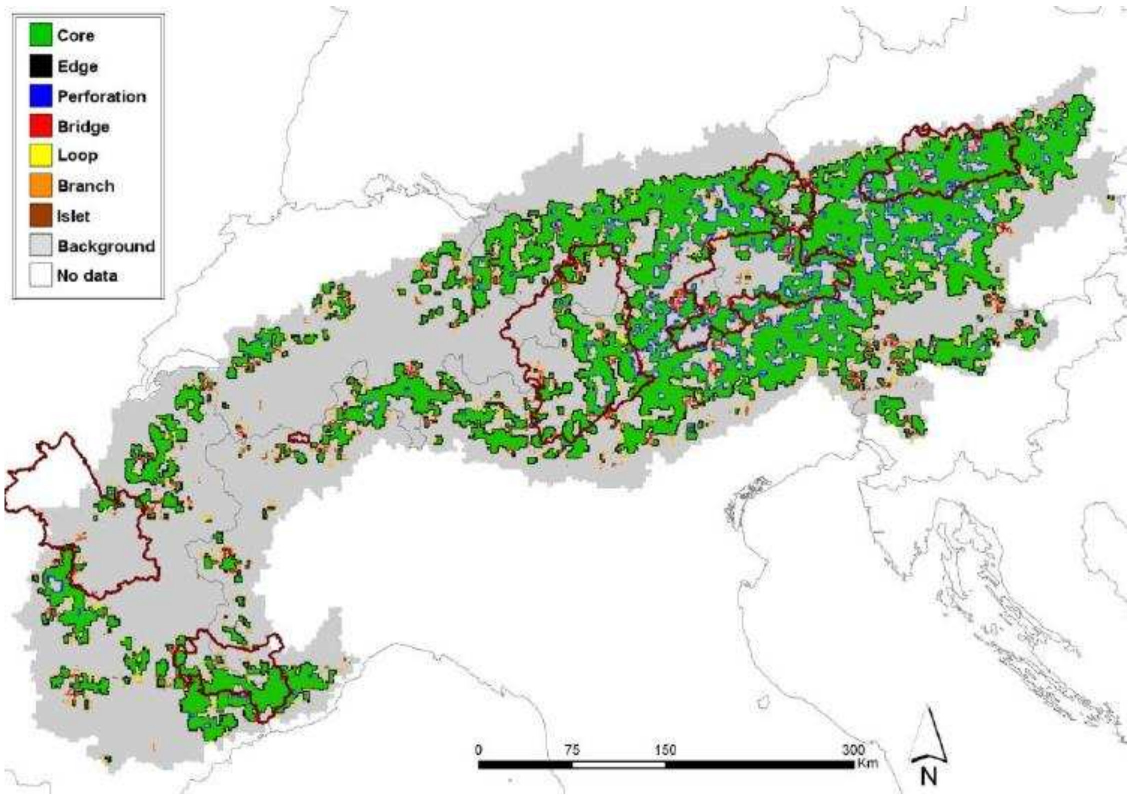


Fig. 7.30b. Morphological spatial pattern analysis based on wolf pack habitat suitability map for the Alpine range. A threshold value of 0.8 was applied. Brown polygons = Econnect pilot regions (Marucco 2011).

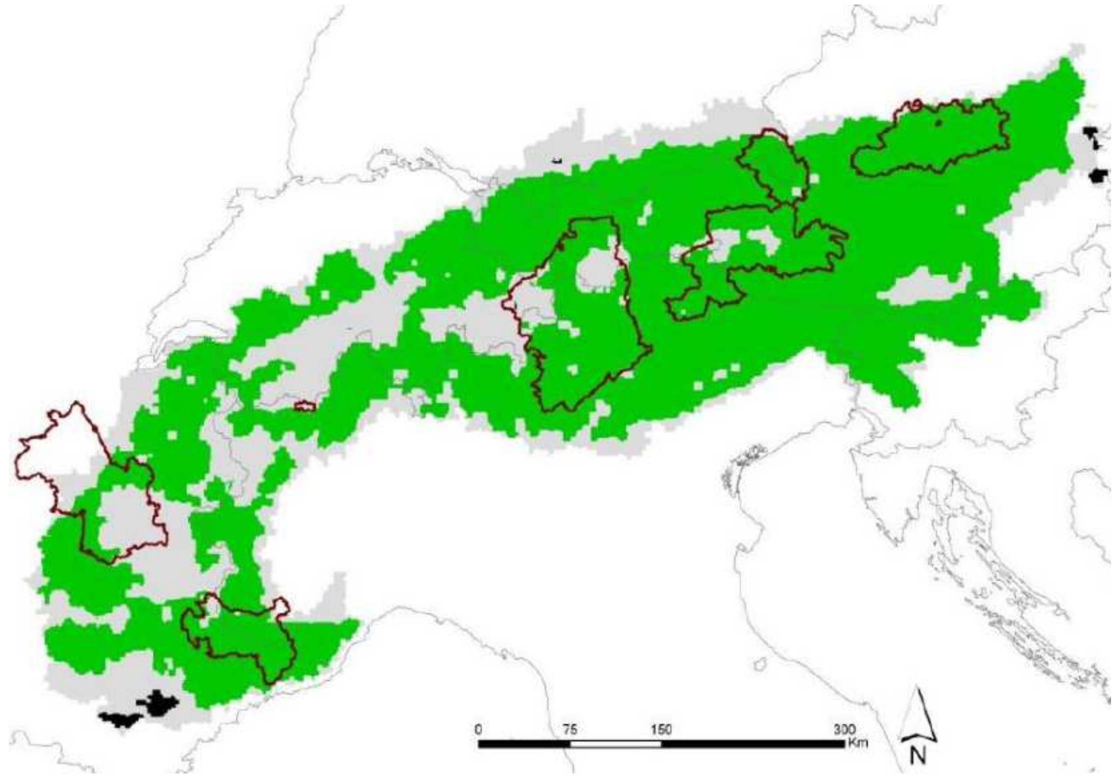


Fig. 7.31a. Network analysis with a threshold of 0.5. Individual components of the network are illustrated with different colours. Brown polygons = Econnect pilot regions (Marucco 2011).

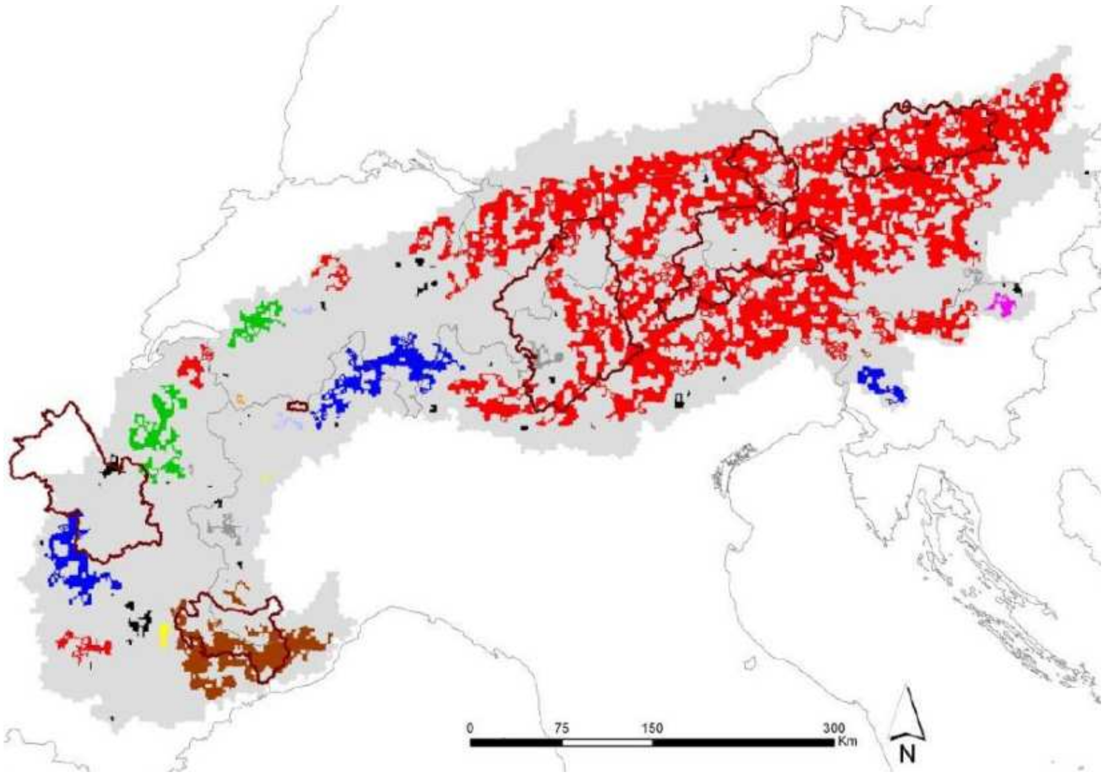


Fig. 7.31b. Network analysis with a threshold of 0.8. Individual components of the network are illustrated with different colours. Brown polygons = Econnect pilot regions (Marucco 2011).

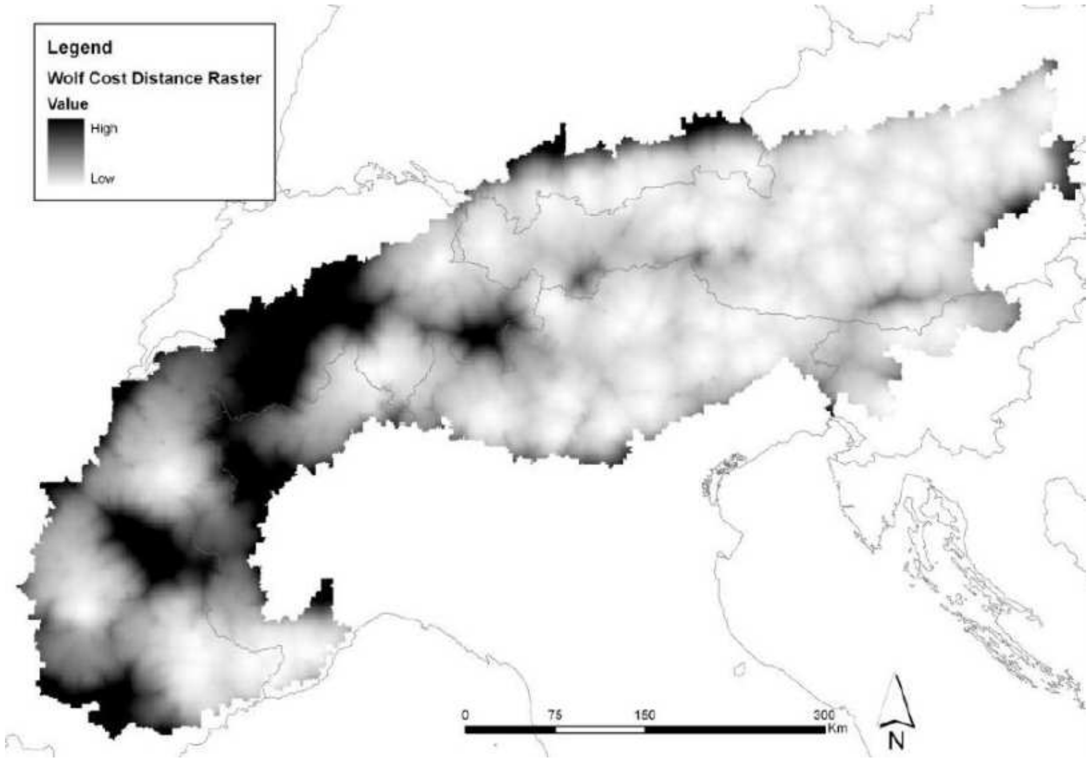


Fig. 7.32. Wolf cost distance raster. For each cell the least accumulative cost distance over a cost surface to the identified source location has been identified (Marucco 2011).

7.3.3. Expected abundance of wolves in the Alps

Alpine Range

Only one wolf abundance estimate for the entire Alpine area is available. Herrmann (2011) estimated a potential number of 1,200–1,580 wolves in the whole Alpine Area based on his habitat model and an assumed wolf density of 1.3–1.7/100 km² (see also Appendix III), as derived from empiric values from the areas with established packs in the French Alps. Such estimation is highly speculative, as it does not consider local differences in prey availability (which are considerable between the western and eastern Alps; Chapter 5.3), but it provides for the countries which have so far no established wolf population a rough and rather conservative first guess (Table 7.2).

France. Herrmann (2011), based on his habitat model and an assumed average wolf density of 1.3–1.7/100 km^2, estimated a potential number of 245–320 wolves in the Alpine Area of France (Table 7.2). In 2012 in the French Alps a minimum of 68 wolves was estimated (Kaczensky et al. 2013a).

Italy. In the Italian Alps, where the landscape is human dominated and suitable habitat is heavily fragmented, habitat constraints on wolf pack establishment can reduce the carrying capacity for wolves (Marucco & McIntire 2010). The number of wolf packs inhabiting the Italian Alps has been predicted to be 36 by 2018 and 49 by 2023 (Marucco 2009, Marucco & McIntire 2010). Pack number estimates and locations are the most useful information to describe wolf expansion, especially for management purposes (Marucco & McIntire 2010). The probability that wolves will establish reproductive packs east of the large lakes in the south-central Alps by 2013 was estimated to be low (but not zero). However, if a pack would establish in the eastern Italian Alps this was considered to drastically change model predictions (Marucco & McIntire 2010). Such an event just took place in 2013 when in the Veneto region, east of the large lakes region, the first reproduction has been confirmed (Parco Naturale Regionale della Lessinia 2013). In 2012, reproduction also took place in the eastern Swiss Alps, when two dispersed wolves from Italy had cubs for the first time.

Herrmann (2011) estimated a potential number of 370–485 wolves in the Alpine Area of Italy based on a MaxEnt model with an assumed density of 1.3–1.7/100 km^2 (Table 7.2). The population estimation for the population for 2010/11 was 70 wolves in 15 packs (Kaczensky et al. 2013a).

Table 7.2. Estimated number of wolves in the Alpine range and parts of the Alps based on the habitat model of Herrmann (2011) assuming a wolf density of 1.3-1.7 individuals per 100 km^2 predicted suitable habitat.

Total area of suitable habitat (km^2)		**Estimated number of wolves**
France	18,875	245-320
Italy	28,520	370-485
Switzerland	12,020	155-205
Liechtenstein	60	<1
Germany	2,610	35-45
Austria	28,880	375-490
Slovenia	1,900	25-35
Alpine range:	92,870	1,200-1,580

Switzerland. Landry (1997b) predicted that the three cantons Valais, Ticino and Grisons could support between 54 and 87 wolves applying a density estimate of 1/50–80 km^2. Considering the number of available prey species (including the number taken by human hunting), the Valais could support 25, the Ticino 54 and the Grisons 94 wolves (Landry 1996).

The model of Herrmann (2011) predicted a population size of around 140–195 wolves for the Swiss Alps, using an assumed density of 1.3–1.7/100 km^2 (Table 7.2). In 2011, 8 single wolves were identified in Switzerland (Kaczensky et al. 2013a). The first pack established in eastern Switzerland in 2012 (Chapter 4.3.1).

Germany. According to the spatially-explicit, predictive rule based models of Fechter & Storch (2014), Germany could host between 154 and 1769 wolf packs or 616–8845 wolves depending on the different model inputs, assuming an average home range size of 200 km^2 and an average pack size of 4–5 individuals (Fechter & Storch 2014). Another study predicted suitable habitat for 400–441 wolf packs in Germany (Knauer et al. unpublished data, in Fechter & Storch 2014). Fechter & Storch (2014) do not give figures for the different regions of Germany. The number of estimated wolves highly depends on the model type, parameters and rules, e.g. if small or isolated patches of suitable habitat are included, the number of wolf packs increases by up to 100% (Fechter & Storch 2014). However, Fechter & Storch (2014) were investigating the uncertainty in rule-based habitat models for habitat generalists, and did not intend to produce a realistic estimation of the potential wolf abundance.

Herrmann (2011) estimated a potential number of 35–45 wolves in the Bavarian Alps based on the estimated suitable habitat and an assumed wolf density of 1.3–1.7/100 km^2 (Table 7.2). No wolves were listed as living permanently in the German Alps by Kaczensky et al. (2013a).

Austria. For the Alpine area in Austria, a potential number of 375–491 wolves were estimated based on the MaxEnt model of Herrmann (2011) and an assumed wolf density of 1.3–1.7/100 km^2 (Table 7.2). The number of wolves in the years 2009–2011 in the Austrian Alps was given as 2–8 (Kaczensky et al. 2013a).

Slovenia. Hermann (2011) estimated a potential number of 25–35 wolves for the Alpine range in Slovenia assuming a density of 1.3–1.7 wolves per 100 km^2 (Table 7.2). Kaczensky et al. (2013a) mentioned only occasional dispersers for the Slovenian Alps.

7.3.4. Hypothetical expansion of the Alpine wolf population

Wolves have a high potential growth rate when they encounter favourable conditions, and if not persecuted, they show high colonisation ability. However, their populations can decline drastically when survival is reduced as they are very sensitive to high killing rates (Chapron et al. 2003).

The wolf population in the Alps is likely to expand mainly from the West to the East (Fig. 7.33a-c; Marucco & McIntire 2010), with the population in the south-western Alps as the main source. Wolf (re-) colonisation takes place in two steps: first young single individuals, mostly young males, sporadically disperse to find new suitable territories and mates. In a second step territories are established and stable reproductive packs are formed if enough suitable habitat is available (Valière et al. 2003, Fabbri et al. 2014). Several years (4–6) can pass between the first arrival of a disperser and the building of a pack (Valière et al. 2003). In the south-western Alps for example, individual wolves were first recorded in the beginning of the 1990s, and the first record of pack establishment was recorded in Italy and France after 1995 (Herrmann 2011).

Wolf colonisation probabilities in the Western Italian Alps were indeed best explained by an additive effect of distance to the closest pack and year, and the proportion of forest habitat (Marucco 2009). From 1999 to 2008 the main source for wolves, which were recolonising the Alpine range, was in the Ligurian-Maritime Alps (Marucco 2009). In the future, the main source will likely be shifted to the north towards the Cozie Alps region from where the recolonisation of the eastern Alps was expected (Marucco 2009). Marucco & McIntire (2010) predicted that from 2009–2023, the wolf pack density will increase in the Italian western Alps and the probabilities of finding a mate will increase. In the eastern Alps, however, wolf density was predicted to remain very low due to low probabilities of finding a mate, even including long-distance dispersals (from various populations) and thus, to slow down recolonisation rate (Marucco & McIntire 2010).

The high dispersal capability of wolves allows for long distance dispersal and solitary wolves were already recorded in the eastern part of the Alps (Fabbri et al. 2014). However, it was suggested that for a recolonisation of the entire Alpine range several wolf packs must be created in the Central Alps acting as a new source for wolf repopulation in the Eastern Alps (Marucco 2009). In Switzerland, the first wolves arrived in the Canton of Valais (western Swiss Alps) in 1995, and the first pack established only in 2012, but remarkably in the eastern Swiss Alps, far away from the “entrance portal”. The establishment of this pack in eastern Switzerland, the establishment of a pack with parents from different source populations in the central Italian Alps (SloWolf 2012; Fig. 7.4 in Chapter 7.1) and the first confirmed wolf reproduction in the Slovenian part of the Alps (WAG 2014) could now be the beginning of the colonisation of the central and eastern Alps. These events also demonstrate how erratic the establishment of new packs can be and how unpredictable the colonisation process is, especially when considering that wolves can immigrate to the Alps from several source populations.

a

b

c

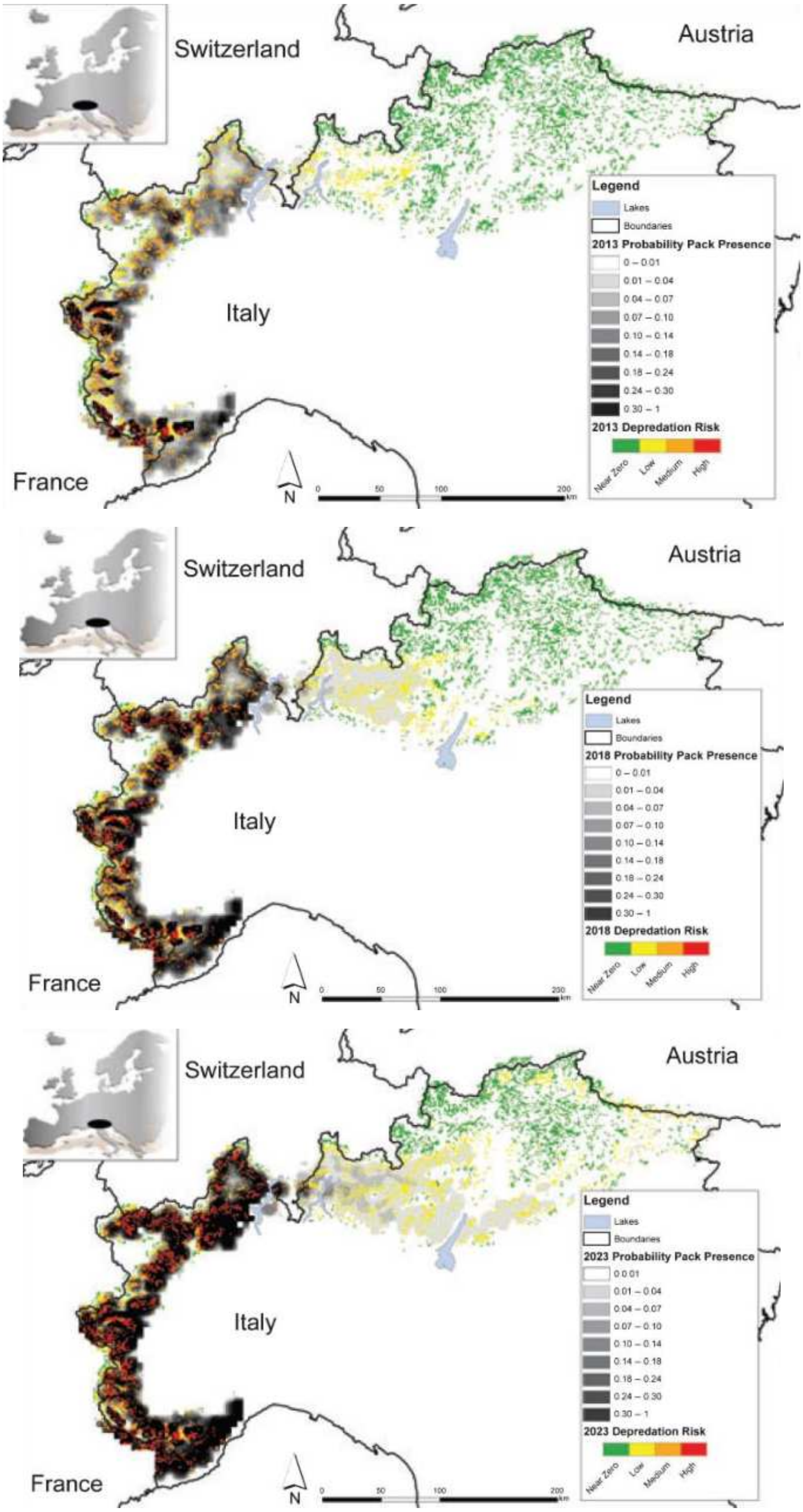


Fig. 7. 33a–c. Absolute probability maps predicting wolf pack locations in a) 2013, b) 2018, c) 2023, based on the SE-IBM of wolf recolonisation process over the Italian Alps. Starting scenario was in 2008. Source: Marucco & McIntire 2010.

7.4. Discussion and conclusions

7.4.1. Lynx

In spite of the different modelling methods, all three lynx habitat suitability models for the area of the Alpine Convention predicted suitable lynx habitat distributed over the entire Alpine range with low suitable areas located mainly in regions with high altitudes (Zimmermann 2004, Signer 2010, Becker 2013). Both, the habitat model of Becker (2013) – using the most up to date lynx data set – and the model developed by Zimmermann (2004), revealed very similar results with regard to the spread of suitable habitat over the entire Alpine range. The model from Becker (2013) resulted in approximately 10% more suitable habitat, but both models indicated a similar distribution of the suitable habitat across the Alps (Zimmermann 2004, Becker 2013). The model of Rüdisser (2001) for western Austria confirmed the other models in so far as that the same areas were classified as suitable or unsuitable. Regarding the amount of predicted suitable habitat, the models from Becker (2013) and Zimmermann (2004) also show a high agreement with the model of Signer (2010). The model of Becker (2013) indicated around 5% more suitable habitat than the model by Signer (2010). However, contrary to the models by Zimmermann (2004) and Becker (2013), which indicated a slightly higher suitability of the western Alps, the model of Signer (2010) indicated a higher suitability for the eastern Alps. This is a consequence of differences between the input data sets (Appendix VI). Zimmermann (2004) and Becker (2013) used among others radio-telemetry data from Switzerland, whereas Signer (2010) based the model on chance observations from Austria. As all habitat models perform better closer to the place of origin of the input data, Zimmermann's and Becker's model may be too conservative for the eastern Alps, while Signer's model may underestimate the suitable habitat in the western Alps.

Different models confirmed the lynx' preference for forested areas, followed by shrubs and herbaceous vegetation and an avoidance of intensive agricultural areas (Schadt 2002, Zimmermann 2004, Basille et al. 2008, Herdtfelder 2012). In the models, lynx avoided urban settlements and areas of high human activity (Zimmermann 2004). However, areas occupied by lynx are not necessarily free of human presence. Distance to roads was not negatively correlated to lynx presence in most of the models of Zimmermann (2004), indicating that, when lynx occur in good habitats, they can adapt to human presence. However, mainly highways seem to affect lynx occurrence (Zimmermann 2004, Basille et al. 2008, Rolland et al. 2011; Chapter 7.2.2). Considering the bias towards the region of origin of data, the fact that Becker (2013) used the most comprehensive lynx data set, and the fact that lynx can adopt to a certain degree to human presence, we assume that at least 100,000 km² in the entire Alpine Arc are suitable habitat for lynx.

The definition of barriers in the model of Becker (2013; Fig. 7.13) was done according to the opinion of KORA team members having worked with lynx. The intention was to define barriers that would hamper the "regular" exchange of individuals and hence (from the point of view of a lynx) make a conspecific living beyond the barrier a "foreigner" compared to lynx living on the same side of the barrier. An individual (dispersing) lynx can overcome almost any obstacle in the Alps (see example of B132 in Fig. 7.1; Chapter 7.1), but strong barriers are clearly hampering the expansion of the population and may in the future also result in genetically distinct subpopulations. However, this hypothesis can only be tested if once the Alps are all occupied and the genetic structure of the entire Alpine population can be studied. Regarding the conservation management of lynx, it is therefore advisable to distinguish several subpopulations across the Alps, e.g. following the suggestion of Becker (2013; Fig. 7.13).

Assuming that the suitable habitat for lynx in the Alps is about 100,000 km² and the density could range from 1.0 lynx/100 km² to 3.0 lynx/100 km², the total population of independent lynx (resident adults and dispersing subadults) would be 1,000–3,000 individuals. This estimation bases on empiric densities observed in Switzerland by means of CMR camera trapping, especially in the north-western Alps (see respective monitoring reports under www.kora.ch). As subadult and adult lynx cannot be properly distinguished on photos, the "number of independent lynx" is used. The effective population (N_e) in lynx corresponds to the "resident lynx", which is equivalent to the "number of mature individuals" e.g. used by IUCN for the Red List assessment. Zimmermann (2004) based his estimations on the density of resident animals as estimated from radio-telemetry studies, and estimated the total number of lynx for the Alps roughly 1,000 (moderate density) to 2,000 (elevated density). As Zimmermann's (2004) model was rather conservative compared to the other models, we assume that this is rather a conservative population estimation, too.

The expansion of the lynx population is very slow; 40 years after the first releases, less than 20% of the suitable habitat in the Alps are occupied. The reasons for the slow expansion are not fully understood, though several factors may play a role: Illegal killing and traffic mortality are the main threats to the survival of individuals and thus the development of the population. However, fluctuations of the population in the north-western Alps and high-density phases did not result in a further expansion, and dispersal and expansion seem not to be density-dependent (Chapter 7.1). Anthropogenic and natural barriers are likely fragmenting the Alps into more than 20 distinct habitat patches or lynx subpopulations (Becker 2013). The low migration rate does not allow for an easy colonisation of such patches, although the rate of exchange of individuals (once neighbouring patches are occupied!) will most likely be sufficient to maintain the genetic viability of all subpopulations and the entire metapopulation – under the condition that the now high degree of inbreeding (Chapter 4.2.2) can be mitigated. Consequently, the remedy of the remnant populations (e.g. the populations north-western Alps and in the Dinaric/south-eastern Alps) and the colonisation of the areas in-between by means of "stepping stones" (the creation of local population nuclei) has been proposed.

7.4.2. Wolf

In spite of the different model characteristics, the results of the various models are mainly in agreement with each other with regard to the main factors influencing wolf presence and distribution. High human density and "disturbance" (roads, settlements) were indicated to negatively impact wolf presence and distribution whereas prey abundance and diversity (where considered), and forested area cover were predicted to have a positive effect (Landry 1996, Massolo & Meriggi 1996, Massolo & Meriggi 1998, Glenz 1999, Marucco 2009, Herrmann 2011, Marucco 2011, Falcucci et al. 2013). High alpine regions are generally indicated as less suitable (Glenz et al. 2001, Herrmann 2011). A preference for multiple-prey habitats may exist because multiple-prey systems possibly provide higher ecosystem stability by reducing high fluctuating prey and predator populations or a multiple-prey system may also minimize man-wolf competition (Massolo & Meriggi 1998).

Another common conclusion of the different studies is that human-caused mortality (traffic accidents, culling, poaching) seems to be the most limiting factor for wolf occurrence and that wolf presence will likely be defined by human pressure and tolerance (Landry 1996, Massolo & Merrigi 1998, Corsi et al. 1999, Glenz et al. 2001, Fechter & Storch 2014). It was suggested that wolves can live even in areas with high road density (indicating high human presence) if they are tolerated and can sustain the traffic-based mortality (Landry 1997b, Fechter & Storch 2014). More than “wilderness” it seems that wolves need sufficient prey and reduced human pressure to survive in a certain region on the long-term (Fechter & Storch 2014).

Predictions based on observations of single wolves colonising new areas resulted in more conservative habitat use than models based on data from established packs (e.g. Herrmann 2011). This is not surprising as we can assume that the first wolves to arrive in a new area would choose the “best” habitat by balancing between high prey availability and little human presence. Wolves prefer to live in such areas, but they can also live somewhere else; as a matter of fact, wolves can live almost *anywhere*. We believe that all these models (and especially the difference between the good and the best habitats) predict rather *how* wolves will occupy the Alps than *where* they may finally live. Wolves are highly adaptable, and the Alps – compared to less mountainous areas presently being recolonisde by wolves – are as a whole very suitable for wolves (Boitani et al. 1998).

Individual wolves (like individual lynx) can overcome almost all linear barriers in the modern landscape, as some spectacular movements of wolves in central and western Europe in recent years have shown. The same is true for the Alps; any point of the Alps can be reached from dispersing wolves from the Italian (or now south-western Alpine), the Dinaric, Carpathian or the Central European Lowland population. Although the spread of the population in the south-western Alps was rather continuous, pairs of wolves can meet and settle down at any site of the Alps and start new population nuclei, as the newly established packs in the Canton of Grisons (eastern Switzerland) and in the Veneto Region (eastern Italian Alps) demonstrated. So neither for demographic nor for genetic reasons is it justified to distinguish subpopulations for the wolf within the Alps. The available habitat models (different to the lynx models) do indeed not suggest such subunits, although Marucco (2011) revealed a higher connectivity in the eastern than in the western Alps. It might however still be practical to distinguish several transboundary wolf regions or compartments beyond or in addition to the “national populations”, which for management reasons will always be relevant.

The suitable habitat for the wolf in the Alps was estimated by Herrmann (2011) to be 49% of the total area or about 93’000 km². (The only other figure was published by Falcucci et al. 2013 with 5.2% suitable area of the Alps, but we believe that this is an error in the paper.) Herrmann (2011) estimated the potential Alpine wolf population to be roughly 1,200–1,500 animals. This may have been speculative, but it indicates that a future Alpine wolf population would be demographically and genetically viable. Indeed, the genetic diversity of the Alpine population will eventually be larger than those of the four European wolf populations that may contribute to the recolonisation of the Alps. Insofar, the Alps as a whole will act as a large corridor for wolf migration.

7.4.3. Model improvement and follow-up work

In two workshops (27 April 2012 in Innsbruck, Austria, and 20-22 August 2012 in Muri, Switzerland), a group of experts from the Alpine countries reviewed the state of the art with regard to habitat suitability models and population viability analyses for large carnivores in the Alps and identified further questions (Chapron et al. 2012). All existing models have their strengths and weaknesses and can shed light on certain, but not all questions. The group concluded that future modelling efforts should build on the existing models and that an approach to produce different models and then compare their outcome would certainly enhance the reliability of the findings. To enhance the prediction power of habitat modelling, a combination of several methodological approaches (e.g.

Autologistic regression, MaxEnt and Mahalanobis distance factor analysis (MADIFA) for lynx and MaxEnt, Ecological Niche Factor Analysis (ENFA), Poisson regression and occupancy modelling for wolf) was recommended for a more detailed assessment of the habitat suitability and population viability. It was not possible to do further modelling within the frame of the project to compile this report. Nevertheless, the proposed approach would be promising and should be considered for further works.

Regardless of the advantages or disadvantages of the approaches used so far, all habitat suitability models agree that sufficient good quality habitat for both lynx and wolf is available in the Alps to host a large population for both species. The populations could be in the range of 1,000 or more animals, and there is no part of the Alps that will not be included in a future distribution area of wolves and lynx. What we lack for the time being – and what could be addressed in future modelling exercise – is a prediction about the spread of the populations and the occupation of new areas across natural or anthropogenic barriers. These are questions that are both more promising and more urgent for the lynx than for the wolf. The lynx, with its rather "obstinate" land tenure system is more predictable than the erratic wolf, and the slow spread and the importance of connecting remnant populations of lynx make such models a priority task.

Only few models have so far included prey densities or prey availability. The simple reason is that such information is not readily available. While a model of the future distribution (habitat suitability) can be done based on habitat, landscape and possibly climatic variables alone, a model describing regional differences in abundance would ultimately have to include prey variables. Answering such questions might be essential with regard to wildlife management when large carnivores are present, but it would require that the Alpine countries or the administrative subunits are generating such data and make information on wild ungulates available in a consistent way.

More targeted and more specific (and hence directly applicable) models could be produced if more precise questions would be asked and more detailed input data would be available. Such information is however, especially with regard to the wild prey populations, often not available (Chapters 5.3 and Chapter 8). The conservation of the Alpine large carnivore populations and the continuous integration of large carnivores into the existing wildlife management systems will bring up a lot of challenging questions which could be tackled, as a first approach, through a modelling exercise.

8. Conclusions

The Alps are a mountain range of almost 200,000 km², of which about half is considered to be suitable habitat for lynx and wolf, distributed over the entire Alpine Arc. The Alps are the largest and most natural landscape in central-western Europe. They could host populations of lynx and wolf of at least 1,000 animals, probably much more. Density estimations for the two large carnivores from the Alps up to now are mainly available from the western part of the range, where the prey abundance in general is lower than in the eastern Alps (see below).

Compared to the period towards the end of the 19th century, when the large carnivores disappeared from the Alps, the current ecological conditions are excellent and we can conclude that the ecological carrying capacity for large carnivores in the Alps is high: The habitat (i.e. forests) has recovered and is now covering 52% of the Alps. The wild ungulates forming the prey base of lynx and wolf – red deer, roe deer, chamois, and wild boar – are more abundant today than in the past centuries, possibly more abundant than ever. There cannot be the slightest doubt that the Alps are an ecoregion that is suited for the presence of lynx and wolf and is, with regard to the ecological conditions, able to host viable populations of these protected species in the future.

The flipside of the coin is that the Alps are also the most intensively used mountain range in the world. Some 15 million people are permanently living in the Alps, and many more are visiting the Alps every year as tourists. Compared to the 19th century, the human population has considerably increased, but the distribution has radically changed. Some regions in the Alps, especially the southern and eastern parts, have seen a net reduction of the human population, and even in regions that experienced a population growth, people are now concentrated in peripheral or inner-Alpine urban centres, in the vicinities of the large cities along the rim of the Alps, along the most important traffic lines or in touristic centres. The average land use has also changed: While in the 19th century, most inhabitants of the Alps were self-sufficient (and made intensive use of all natural resources), the majority of people living permanently or temporarily in the Alps lead a rather "urban" live and use Alpine landscapes mostly for recreational reasons. The tendencies toward a concentration of people in the (peripheral) centres, the depletion of remote areas without tourism and the further decrease of the traditional economy of the Alps (agriculture, livestock breeding) is prognosticated to continue over the coming decades (Bätzing 2013).

The concentration of the human population along the main valleys and creation of modern transport infrastructure led to an increased artificial fragmentation of a landscape that is (for mainly forest-living species) naturally fragmented through high alpine ridges. While this fragmentation is no problem for the movements of individuals (although an increased mortality risk through traffic accidents exists), it is obviously obstructing the expansion of the lynx population(s) and hence hampering the natural recolonisation of the Alps. For the wolf (which can found new population nuclei detached from a permanently occupied area), the intra-Alpine fragmentation is considered a minor problem for the expansion of the population.

The really big challenges for the survival of lynx and wolf in the Alps are not the ecological conditions, but the acceptance by people and the question of how coexistence between traditional land users and the returning large carnivores can be organised. There is no consensus about the return of the large carnivores in the societies of the Alpine countries. The only obvious conclusion so far is that the reintegration of large carnivores into cultivated landscapes (hence also the Alps) needs to be *managed*. A *laissez-faire* policy will either prevent the establishment of vital populations or it will result in enduring conflicts that our democratic societies should not be willing to accept. Most countries have realised the need for certain rules with regard to the management of large carnivores and have drafted and implemented management plans (Chapter 6). We have listed these plans in this report, but it was not possible within the frame of this project to make a comprehensive review and assessment. In order to harmonise the approach to large carnivore conservation management across the Alps, one first assignment would be to compare the existing management plans.

The question of tolerance of people towards large carnivores has been addressed in two different reports (Mondini & Hunziker 2013; Mikschl et al. 2014). Here, we merely discuss the technical aspects of the challenges of coexistence between large carnivores and people.

The *first* aspect to consider is the fear people have of large carnivores. In the case of lynx and wolf (and different to the brown bear), this fear is objectively not justified, as the risk that a wolf or a lynx would attack a person is extremely low. In no region with autochthonous wolf or lynx populations, people consider them to be dangerous. We can assume that eventually, people living in the Alps will realise this again, too (as is already the case in regions where these species have been present for a while). Nevertheless, the fear of people in all areas where these species arrive is a serious issue that needs to be addressed. Large carnivores have a plastic behaviour and are highly adaptable. As a consequence, they can learn to cope with human presence and human activities. Especially the intentional or repeated accidental offering of food can quickly habituate wolves and bears (less lynx, as they normally do not scavenge) and considerably increase conflict with people and (especially for the bear) increase the risk of accidents. As experience has shown that habituated individuals are hard or even impossible to re-educate, habituation should by all means be avoided. We recommend that the Alpine countries should find a consensus on how to deal with wolves and lynx that have "lost their shyness" towards people. And first, we would need to study such cases to be able to draw the line between "normal" and "habituated" behaviour. This is a much more serious problem with bears and is presently addressed by the pilot action "Defining, preventing, and reacting to problem bear behaviour in the Alpine region" in the frame of an EU project on large carnivores (Majić Skrbinšek & Krofel 2014). Nevertheless, the question of habituation should also be addressed for wolf and lynx.

The *second* and more important and lasting issue are the attacks by large carnivores on livestock, mainly sheep. The approaches so far to solve this conflict are threefold: (1) preventive measures (shepherds, electric fencing at night, livestock guardian dogs), (2) compensation of confirmed losses, and (3) removal of notorious stock raiders. While the first two measures are applied (or are foreseen to be applied) in all Alpine countries, the legalised lethal removal of problem animals is more controversial and has so far been applied for wolves or lynx only in France, Switzerland, and Slovenia. Lethal control of stock raiders is clearly correlated to the magnitude of attacks and the political pressure from people and institutions concerned.

The transition from sheep husbandry practices without large carnivores (where sheep were left unattended freely grazing on subalpine and alpine pastures most of the summer) to a guarded husbandry practice requiring personnel and hence a considerable financial investment is difficult and meets the fundamental opposition of most sheep owners in the regions concerned. In some Alpine countries, financial incentives are provided by the governments to facilitate this transition, e.g. guarded sheep flocks get higher subsidies than un-guarded. The preventive measures taken so far are more or less the same as those applied in regions with autochthonous large carnivore populations. These experiences show that it is in principle possible to have a lively livestock husbandry system even if large carnivores are present. The re-adaptation will however take time (at least several decades), and it will require a more professional approach to sheep breeding in the countries where over the past 50 years, keeping sheep has turned into a side job or a mere hobby. A sheep husbandry system with a much tighter surveillance of the animals than so far does not only allow reducing wolf attacks, it brings also additional advantages, e.g. a better control of transmission of diseases among domestic animals or between livestock and wildlife, and a more sustainable grazing of the fragile alpine pastures. This implies that sheep husbandry in the Alps is considered in a broader context than only with regard to losses to large carnivores. This might be less a question of international than of national cooperation, e.g. between authorities responsible for nature conservation, for wildlife management, and for mountain agriculture and livestock health. It might however be helpful to discuss the (economic) significance of sheep husbandry in high mountain areas and its relation to other conservation or economic goals in a broader that is in an international context.

An obvious topic for pan-Alpine cooperation is the exchange of experience with attacks on livestock and with preventive measures. As far as we can see, the (international) contacts are working well among experts, but a lot of the data compiled on livestock losses for this report were rather cryptic and not readily available. Some national or regional agencies seem to treat information on livestock losses and compensation as classified data (or were not able to produce the data requested). With regard to an international cooperation and sound analyses, the availability of information should be improved.

The *third* topic is the impact of wolf and lynx on wild prey populations and the concern of hunters that the presence of large carnivores will substantially reduce their harvest. This has been much less discussed so far than the attacks of large carnivores on livestock, but it is, to our understanding, the most crucial question for the return and the survival of large carnivores in the Alps and should be given much more attention in the years to come. With regard to the lynx, where attacks on livestock are a minor issue and relatively easy to manage, the conflict with hunters over the reduction of roe deer and chamois populations are by far the greatest challenge, and illegal killing is – not only in the Alpine countries – considered the number one threat. Lynx can indeed show a remarkable numeric response to improved prey availability and can subsequently considerably reduce prey abundance. Such situations in the Swiss Alps have inevitably triggered severe conflicts with hunters and have always resulted in increased illegal killings. The experiences from Switzerland have shown that the temporary strong impact of lynx on roe deer (less obvious on chamois) never occurred independently from other influences such as (changing) winter mortality or hunting pressure; this however just illustrates that lynx predation needs to be considered in wildlife management decisions.

There is – to our knowledge – no such experience with regard to wolf predation in the Alps. Wolves were reported to have had a strong influence on mouflon in the Mercantour and subsequently showed a temporarily increased predation on chamois. But generally, the reduction of wild ungulate populations by wolves seems not (yet) to be a strong issue in the Alps. This might be because of lack of information or because wolves did not reduce prey abundance. Indeed, wolves (compared to lynx) may cause more compensatory mortality or show a stronger and faster functional response to changes in prey availability as they have a wider prey spectrum and are scavengers.

But predation and predation impact may also depend on the region within the Alps. The wolf has so far established a permanent population in the south-western Alps of France and Italy, and observations on lynx predation came predominantly from the north-western Alps in Switzerland.

Experiences pertaining to large carnivores and wild ungulates from the eastern Alps are so far not available (with exception of Slovenia, but there mostly from the southern part of the country).

There is a remarkable difference in wild ungulate harvest between the western and the eastern Alps (Fig. 8.1). Average harvest densities in eastern Switzerland, Liechtenstein and further east are much higher than in the western parts (with exception of Slovenia with most of the area considered in Fig. 8.1 outside the Alps). We can, at the moment, not assess the differences of the impact related to the prey density (assuming that the harvest density indeed is representative for the population density). It could be that increased prey availability will result in a considerably higher wolf and lynx density in the eastern Alps; it could however also mean that the impact on the wild prey populations is smaller than in the western Alps, if predator densities are not directly correlated to prey density or availability.

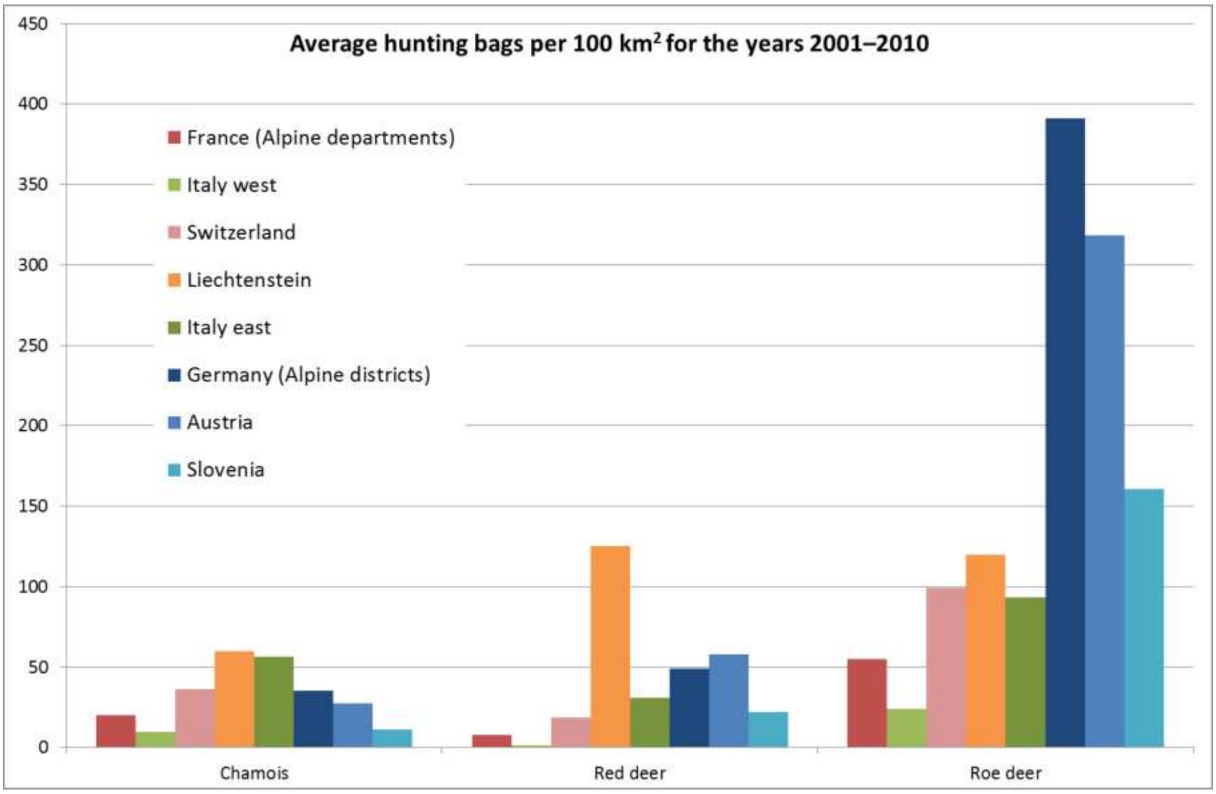


Fig. 8.1. Comparison of the average hunting bags per 100 km² for the years 2001–2010, sorted from west to east. Italy west: chamois: data from the provinces Imperia & Lecco 2003–2010; red deer: Lecco 2009–2013; Roe deer: Lecco 2003–2010. Italy east: chamois: Bozen; red deer: Bozen & Treviso; roe deer: Bozen & Treviso. Sources: FR: Data "Réseau Ongulés Sauvages", ONCFS/FNC/FDC ; IT east: G. Torello, pers. comm., R. Facoetti, pers. comm.; CH: BAFU 2014; FL: Wolfgang Kersting, Amt für Umwelt, Liechtenstein pers. comm.; IT west: M. Stadler, pers. comm., S. Busatta, pers. comm.; DE: Reinhard Menzel, Bayerisches Staatsministerium für Ernährung, Landwirtschaft und Forsten pers. comm., Friedrich Pielok, , Bayerisches Staatsministerium für Ernährung, Landwirtschaft und Forsten pers. comm., Frank Tottewitz, Thünen-Institut für Waldökosysteme, pers. comm.; AT: Statistik Austria 2014; SL: Statistical Office of the Republic of Slovenia 2014.

A discussion on possible predation impact is at this point rather speculative, but the facts behind the west-east gradient shown in Fig. 8.1 raise a few questions and points to consider:

1. The data base on the wild ungulates across the Alps is not good. The only rather consistently available data sets were the hunting bags, and even those we received only from 4 out of the 24 contacted provinces of the Italian Alps. It is unclear how representative hunting bags are for the local

ungulate populations, but other data – if ever available – are even less consistent and hence more difficult to compare. For a meaningful integration of large carnivore predation into wildlife management practices, the knowledge base for the prey species should be improved, not only to allow international comparisons, but also to understand the local predation impact in relation to other mortalities.

2. The increased densities of wild ungulates in the eastern Alps reflect differences in the wildlife management practices and the (economic) importance of hunting on a regional level. The high level of large herbivore densities in the eastern Alps are the consequence of and the reason for some particular wildlife management practices which are not applied in the western Alps, such as extensive feeding and winter enclosures for red deer. We can assume that such practices will cause particular situations when the large carnivores return. Such aspects need to be considered mainly on the national level, but the consequence is that (1) experiences from the western Alps may not be directly applicable in the eastern Alps, and (2) the return of the large carnivores in the eastern Alps will not only influence the wildlife management practices, but also the forestry practices, which are more strongly connected to game management than in the western Alps.

3. The dialog with hunters *and* foresters should be strengthened. Although the changes in livestock husbandry provoked by the return of the large carnivores are more visible (and more covered by the media), the adaptations needed in wildlife management (and possibly forestry) will be as essential and as challenging and require adequate partners.

4. The differences in wildlife management and hunting systems between the Alpine countries are much stronger than with regard to people's attitudes or livestock/sheep husbandry. The response of local people or of sheep owners to the return of the large carnivores across the Alps are variations of a common theme. Regarding the status of game populations and wildlife management traditions, we see fundamental differences. The international dialogue, let alone a harmonisation of wildlife management approaches to large carnivores will hence be much more challenging. Consequently, it would here be even more important to define common and over-arching goals and then to try to achieve these goals in each country under the respective national/regional preconditions.

Such principles – common goals for an entire population and adaptation of targets and procedures at the national or regional level – are the basic ideas behind the "Guidelines for population level management plans for large carnivores" (Linnell et al. 2008). The objective of Working Group 3 of the RowAlps Project is to outline possible management scenarios and ways of cooperation for the conservation and management of the Alpine lynx and wolf populations. Although this may not immediately result in a "Pan-Alpine Management Plan", the suggestions of the Guidelines should be considered, especially as they provide a pragmatic approach to the challenge of conservation and management of large carnivores in a cultivated landscape.

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Appendix I. MALME

MALME
Metapopulation Approach for large Mammals in Europe - Case Study Alps

Background

Nature protection legislation is based mainly on species protection and upon protected areas. Neither approach is practical for the conservation and management of species which require vast areas to form viable populations and that have different conservation status among the regions. This is typically true for large carnivore species like the brown bear, wolf and lynx, but also for other large mammal species whose populations extend far beyond protected areas and whose presence however causes conflicts with human land uses. For all these species a differentiated concept that sets the focus on the (meta) population would make more sense, both in regard to the biological/ecological needs of the species as well as for differentiated management aspects. Such an approach however requires a comprehensive proceeding which takes into account the different management and conservation concepts with their varying often contradicting priorities.

A broad partnership between state institutions, experts and interest groups, a strong conceptual framework and – as a working tool – a network of information is needed to achieve this.

The MALME or Metapopulation Approach for Large Mammals in Europe is an open access platform on the internet which not only contains peer reviewed papers but also data pertaining to land use and approaches aimed at solving current conservation problems in the Alpine arc. It aims at giving a thorough overview of all aspects of wildlife conservation in the Alps. Almost all information except for the documents is available in English, French, German and Italian.

The main headings available include:

- Information & News
- Species (detailed information on natural history of key Alpine species)
- Land use & management
- Policy
- Library (compilation of peer reviewed scientific papers)
- Maps (species distribution)
- Statistics & Institutions (statistics for land use, carnivores and herbivores; GOs and NGOs)

http://www.kora.ch/malme/20_malme/home/index_en.htm

Also accessible through:

http://www.kora.ch/index.php?id=111&L=1

Appendix II. Threats to the survival of lynx and wolves in the Alps and in Europe

Large carnivores are on the rise again in Europe. Approximately one third of mainland Europe hosts at least one of the four large carnivores: lynx, wolf, bear (*Ursus arctos*) or wolverine (*Gulo gulo*; Chapron et al. 2014). Most populations show stable or increasing numbers for the 21st century (Chapron et al. 2014). However, the survival of these populations is still not secured as they may be facing serious threats.

The threats listed in Table II–1 (lynx) and Table II–2 (wolf) were compiled from reports that have been published since 2000, regarding either the Alpine population or the European population in general.

Lynx

The reports published since 2000 largely agree on the main threats to the lynx population in the Alps and in Europe in general. They consist mainly of persecution, accidental mortality (vehicle collisions), habitat deterioration due to infrastructure development, and low acceptance due to conflicts with hunters, combined with the intrinsic limited dispersal capability of the species. The more recent assessments also list inbreeding as a threat for the present and/or future lynx population. Additionally, the most recent assessment in Boitani et al. (2015), lists poor management structures as a threat to the present lynx population in Europe. The same was already the case in Kaczensky et al. (2013a), the analysis of which was performed by sending questionnaires to the members of the Large Carnivore Initiative for Europe (LCIE) and further experts, for the present lynx population in Europe. However, poor management structures were not among the top 4 threats for the future population. This is due to the increased importance of other threats, not due to a decrease in importance of this threat (Table II–3).

Wolf

The four assessments that have been published since 2000 all list human caused mortalities as one of the main threats, be it shooting, hunting, poaching, persecution, poisoning, accidental mortality, or vehicle and train collision. No other element is listed as a threat to the whole Alpine population in KORA (2015), but some further elements are listed as threats in either France, Italy or Switzerland. Boitani (2000), Kaczensky et al. (2013a), and Boitani et al. (2015) all list low acceptance and poor management structures as main threats. Additionally, Kaczensky et al. (2013a; Table II–3), and Boitani et al. (2015) list the deterioration of habitat due to infrastructure development as a main threat to the European population.

Table II–1. Threats to the Eurasian lynx, compiled from reports published since 2000. Sources: 2000: Breitenmoser et al. 2000; 2001: von Arx et al. 2004; 2003: Molinari-Jobin et al. 2003; 2011: Council of Europe 2012; 2013: Kaczensky et al. 2013a; 2015: Boitani et al. 2015

Year	Area	Time[1]	Threats
2000	Europe	Present	- Deterioration of habitat and prey base - Direct human caused mortality (intentional and accidental) - Diseases, demographic and genetic factors (little evidence) - Sources of conflicts and negative human attitudes (hunters)
2001	Alps[2]	Present	- Infrastructure development: human settlement - Infrastructure development: road building - Shooting (illegal) - Poisoning - Vehicle and train collision - Avalanches/landslides - Competitors & Prey/food base & Pathogens/parasites - Limited dispersal - Low densities
2003[3]	Alps	Present	- Methodological: Ineffective monitoring in AT, IT, SL & DE - Biological: Limited expansion capacity - Biological: Potentially inbreeding - Anthropogenic/management: conflicts – perception of lynx by people - Anthropogenic/management: conflicts – suspected (not necessarily true) impact on hunted wild ungulates - Anthropogenic/management: conflicts – presence and magnitude of predation on livestock
2011	Europe	Past	- Infrastructure development – Transport (land) - Accidental mortality – vehicle collision - Persecution – other - Intrinsic factors – limited dispersal - Lack of public acceptance – low acceptance due to conflicts with livestock - Lack of public acceptance – low acceptance due to conflicts with hunters
		Present	- Infrastructure development – Transport (land) - Accidental mortality – vehicle collision - Persecution – other - Intrinsic factors – limited dispersal - Lack of public acceptance – low acceptance due to conflicts with hunters
		Future	- Infrastructure development – Transport (land) - Accidental mortality – vehicle collision - Persecution – other - Intrinsic factors – limited dispersal - Intrinsic factors – poor recruitment / reproduction / regeneration - Intrinsic factors – inbreeding - Lack of public acceptance – low acceptance due to conflicts with hunters

[1] Past, Present, or Future, relative to the year of the publication
[2] Additionally available for the individual countries (CH, SL, IT, AT) and Europe
[3] Assessed “Problems“, not “Threats“

Year	Area	Time[1]	Threats
2013	Alps[4]	Present	- Persecution - Low acceptance due to conflicts with hunters - Infrastructure development due to Transport (roads/railways) - Inbreeding
	Europe	Past	- Low acceptance - Persecution - Accidental mortality - Poor management structures
		Present	- Low acceptance - Persecution - Accidental mortality - Poor management structures - Habitat (Infrastructure)
		Future	- Low acceptance - Persecution - Accidental mortality - Habitat (Infrastructure)
2015	Europe	Present	- Low acceptance (largely due to conflict with hunters) - Persecution - Habitat loss due to infrastructure development - Poor management structures - Accidental mortality

[4] Additionally available for the individual countries

Table II–2. Threats to the wolf, compiled from reports published since 2000. Sources: 2000: Boitani 2000; 2005: KORA 2015; 2013: Kaczensky et al. 2013a; 2015: Boitani et al. 2015

Year	**Area**	**Time**[5]	**Threats**
2000	Europe	Present	- Hunting and poaching - Habitat quality (lack of retreat areas) and food availability (shortage through hunting purely from hunters' perspective) - Small number, low densities and demographic fluctuation - Range shape and fragmentation (narrow, elongated and fragmented shapes) - Genetic identity (although variability seems normal and hybrids do not incorporate into wolf populations) - Legislation - Management authority (fragmentation) - National and sub-national management (often just national "umbrella" and sub-national management plan not adequate in temporal and spatial scales) - Law enforcement (lack of enforcement) - Economic conflicts (damage to livestock) - Public opinion
2005	Alps[6]	Present	- Shooting - Poisoning - Vehicle and train collision
2013	Alps[7]	Present	- Low acceptance - Selective logging - Poaching - Poor management structures
	Europe	Past	- Low acceptance - Poor management structures - Persecution - Accidental mortality
		Present	- Low acceptance - Poor management structures - Persecution - Habitat (Infrastructure)
		Future	- Low acceptance - Accidental mortality - Persecution - Habitat (Infrastructure)
2015	Europe	Present	- Low acceptance - Habitat loss due to infrastructure development - Persecution - Hybridization with dogs - Poor management structures - Accidental mortality

[5] Past, Present, or Future, relative to the year of the publication
[6] Additionally available for the individual countries
[7] Additionally available for the individual countries

Table II–3. Threat assessment over all populations in Europe based on questionnaires with threats grouped in 19 main categories. For the lynx, 22 questionnaires were filled in, and 28 for the wolf. The number in the table indicates how many times an issue was ticked as a threat (Kaczensky et al. 2013). The list was sorted for present threats for lynx.

Threat category	**Lynx (N = 22)**			**Wolf (N = 28)**		
	Past	Present	Future	Past	Present	Future
Low acceptance	20	20	20	27	27	27
Persecution	16	17	19	19	22	23
Habitat (Infrastructure)	11	16	18	15	22	23
Accidental Mortality	15	16	18	18	20	24
Poor management structures	15	16	16	21	22	21
Lack of knowledge	12	15	12	16	18	13
Intrinsic factors	13	14	16	10	12	14
Change in native fauna	9	13	14	13	15	17
Disturbance	9	12	13	12	14	16
Habitat (Forestry)	8	10	10	8	8	9
Prey over harvest	7	9	10	9	8	12
Habitat (Livestock)	5	5	6	10	10	9
Habitat (Divers)	5	5	5	5	8	6
Harvest	5	4	5	9	12	16
Natural disaster	3	4	4	4	4	5
Habitat (Agriculture)	1	2	2	3	5	5
Habitat (Mining)	2	2	2	4	5	5
Invasive alien Species	1	2	3	4	5	7
Pollution	2	2	3	2	2	2

Appendix III. Density and territory size in lynx and wolf

We present an overview of published densities and territory sizes of lynx (Table III–1 & III–2) and wolf (Table III–3 & III–4). Density is defined as the number of animals per area unit. In the case of large carnivores with extended individual home ranges, 100 km² is often used as reference area. Density is a very simple parameter to compare the status of populations, albeit hard to actually measure in the field. Density data need further information to be understood and comparable. For example, over-all population density can vary over the year. Therefore it must be declared (1) during which season a given density was estimated (e.g. early winter, late winter). Furthermore, it must be stated, (2) which animals were included in the estimation: age classes (adults, subadults, juveniles) and social classes (e.g. pack members, resident animals, dispersers, floaters). Finally, (3) the reference area needs to be defined, e.g. whether the estimations refers to the total area or to the suitable habitat, forested area, for the "permanently occupied area" or including the "sporadically occupied area", etc.

Similar issues exist with "territory size". There are figures published e.g. for the total range, the home range, or the kernel area (or any other statistical model) of an individual. These differ in the exclusion of outliers, the weighting of observation clusters, and the calculation of the polygon(s) of the area. The difference between these ranges can be considerable. A study about the wolves in Germany found that the exclusion of the 5% of GPS positions, which diverged most from the rest, almost halved the calculated territory size (Reinhardt & Kluth in prep.). As the term *home range* in that sense has a very clear definition, we use here *territory size* as a more general term, but some studies also use *home range* in a more general sense, which can lead to confusion. The territory size gives a value for the space used by an individual or a social group, but not about the number of animals within that area, as home ranges may overlap (e.g. home ranges of males and females in lynx), or more than one animal can share this home range (e.g. a wolf pack of potentially unknown size).

There are a lot of different methods to gather information about animals in the field, ranging from systematic monitoring of animals by means of radio telemetry to gathering and assessing sightings and signs reported by e.g. hunters, foresters or the general public. These methods differ in their scientific robustness and accuracy. When comparing data from different studies, the field method must be considered as much as the models used to calculate the territory size.

Unfortunately, findings are not always presented along with the necessary information. When authors present a compilation of data from various studies, regions or countries, they often only give the actual number without additional information – which is, as a matter of fact, often also incomplete in the original publications. Therefore, different densities and average territory sizes without additional information can only be compared with caution!

General correlations between density, territory size and prey

Territory size and density are correlated because territory size is density dependent. Scientists have found correlations between prey abundance and density or territory size for lynx, wolf and carnivores in general. Carbone & Gittleman (2002) found "remarkable consistency in the average population density in relation to prey biomass and carnivore mass". A prey mass of 10,000 kg can support about 90 kg of carnivore biomass. Deviations from this ratio can be caused e.g. by competition with other carnivores. The largest deviation from this rule was found in the Eurasian lynx, which was rare relative to the estimated prey biomass. This might have been caused by illegal killing, known to have occurred in the population where the data came from (Carbone & Gittleman 2002). The general correlation of densities of carnivores and their prey also explains the differences in carnivore density and territory size on a continental scale. The lower environmental productivity towards the Polar Regions can generally sustain fewer herbivores per area and *ergo* also fewer carnivores.

Lynx. In Europe, the largest home ranges for lynx were found in Scandinavia (Breitenmoser et al. 2000). The correlation between lynx home range size and prey density has been found e.g. by Herfindal et al. (2005), who used hunting statistics to calculate the correlation. Additionally, they found that male lynx increased their home range size more rapidly than females as a response to a decrease in prey density (Herfindal et al. 2005). Molinari-Jobin et al. (2007) analysed, which factors predicted best the home range size in different regions of Switzerland. The primary factors differed between territory sizes based on minimum convex polygon (MCP) and Kernel. Variations in the MCP territory sizes were best explained by the interactions of the study (most likely variations in the population status, hence local abundance) with the number of locations per lynx, the number of roe deer harvested per km² and the occurrence of good roe deer habitat; variations in Kernel territory sizes were best explained by the interactions of the study with the occurrence of good roe deer habitat, the occurrence of good chamois habitat and the interaction of good roe deer and chamois habitat plus an additive effect of study (see above). In other words, the manner of the correlation was more complex than suggested by the study of Herfindal et al. (2005) in Norway, and hunting statistics did not offer a simple correlation with territory size, while good habitat was the better predictor. Harvest numbers added explanatory power to the variation in MCP territory size, but nor for Kernel. "*While hunting statistics are not good predictors for the present ungulate population density, they can, however, indicate the population trend and allow for relative comparisons between the present studies*" (Molinari-Jobin et al. 2007).

Wolf. Wolves have the largest home ranges within North America in the northern parts of the continent (Fuller et al. 2003). The same pattern was found in Europe. Territory sizes in south-central Europe were 80-240 km², and 415-500 km² in northern Scandinavia (Okarma et al. 1998). Various authors mention the correlation between wolf density / territory size and prey density (e.g. Apollonio et al. 2004; Ballard et al. 1997; Fuller 1989 cited in Larivière et al. 2000). The general picture is that high ungulate densities result in smaller wolf territories and increased wolf densities (Fuller 1989 cited in Larivière et al. 2000). However, there are complications to this general picture. Variations in this correlation may simply result from methodological issues. Usually, the longer the radio-tracking observation, the larger the territory size. Six to eight months of intensive radio-tracking is the minimum time span for a reliable estimation of the size of a territory size (Okarma et al. 1998). Furthermore, there are also ecological reasons for variations in the correlation. For example, wolf densities in Northwest Alaska would have the potential to be twice as high according to the ungulate biomass indices. However, due to the nomadic behaviour of the caribou, the main prey is not available to most wolves all year long (Ballard et al. 1997). The species of main prey also has an influence. Fuller et al. (2003) found in North America on average much larger territory sizes where moose constituted the main prey (817 km²) compared to where deer was the main prey (199 km²), despite that both areas had similar prey biomass. Fuller et al.'s (2003) explanation was that moose are simply harder to catch for wolves, i.e. the actual availability of the prey biomass to wolves differs between different prey species. A similar notion is expressed by Apollonio et al. (2004):

"[T]wo tendencies are seen. In northern regions, where moose and caribou are major prey, wolves gather in large packs (on average 4–10 members), defend vast territories, their density is low (0.1–2.0 per 100 km²) and they may become nomadic. On the contrary, at lower latitudes, where wolves prey mostly upon deer and wild pigs, they live in small packs (on average 3–6 members), occupy smaller territories and may reach high values of density (2.0–6.0 per 100 km²), depending on prey availability and level of harvesting" (Apollonio et al 2004).

They conclude that local wolf density in their study area (Casentinesi Forest, northern Apennines, Italy) is mostly influenced by the number of wolf packs rather than by pack size (Apollonio et al. 2004).

Table III–1. Densities of Eurasian lynx, calculated / estimated from scientifically robust field methods. Comments for density include additional information, e.g. season, or animals included. Ind. = Independent. Field methods: CT = camera trapping; RT = radio-telemetry; TC = track counts, e.g. snow tracking. Table sorted alphabetically by population/country.

Area	**Density [ind/100 km²]**	**Comments**	**Field**	**Source**
France (Ain)	1.6 (1.2-2.0)	Ind. lynx	CT	ONCFS 2014b
France (Doubs)	0.9 (0.7-1.1)	Ind. lynx	CT	ONCFS 2014b
France (Jura - Ain)	1 (0.7-1.4)	Ind. lynx	CT	ONCFS 2014b
FYR Macedonia (Mavrovo) 2008	0.84±0.24	Ind. lynx per 100 km² suitable habitat	CT	Stojanov et al. 2013
FYR Macedonia (Mavrovo) 2010	0.80±0.31	Ind. lynx per 100 km² suitable habitat	CT	Stojanov et al. 2013
FYR Macedonia (Mavrovo) 2013	1.57±0.32	Ind. lynx per 100 km² suitable habitat	CT	Stojanov et al. 2013
Poland / Belarus (Białowieża P. F.)	1.9-3.2	Winter density, only adults	RT, TC	Jędrzejewski et al. 1996
Poland / Belarus (Białowieża P. F.)	2.8-5.2	Winter density, incl. kittens	RT, TC	Jędrzejewski et al. 1996
Switzerland (8 study areas)	0.92-3.61	Ind. lynx per 100 km² suitable habitat	CT	KORA 2014
Switzerland (Northwestern Alps)	1.9-2.1	Ind. lynx per 100 km² suitable habitat	RT	Breitenmoser-Würsten et al. 2001
Switzerland (Northwestern Alps)	1.4-1.5	Resident lynx per 100 km² suitable habitat	RT	Breitenmoser-Würsten et al. 2001

Table III–2. Territory sizes of Eurasian lynx, calculated / estimated from scientifically robust field methods. Territory size: m = males; f = females. Comments: MCP = minimum convex polygon. Field methods: RT = radiotelemetry; TC = track counts, e.g. snow tracking. Table sorted alphabetically by population/country.

Area	**Territory size [km²]**	**Comments**	**Field**	**Source**
France/Switzerland (Jura)	m: 226 (110–328) f: 119 (62–224)	95% Kernel	RT	Breitenmoser-Würsten et al. 2007
Norway (Akershus)	m: 812 f: 350	100% MCP	RT	Herfindal et al. 2005
Norway (Hedmark)	m: 886±356 f: 535±225	95% Kernel	RT	Linnell et al. 2001
Norway (Nord-Trøndelag)	m: 1499±944 f: 610±85	95% Kernel	RT	Linnell et al. 2001
Norway (Nord-Trøndelag)	m: 1719±252 f: 235±36	95% Kernel	RT	Sunde et al. 2000
Poland / Belarus (Białowieża P. F.)	m: 194±70 f: 100±42	100% MCP	RT, TC	Jędrzejewski et al. 1996
Poland (Białowieża P. F.)	adult m: 235±52 adult f: 152±37 subadult m: 310±165 subadult f: 95±41	95% Kernel	RT	Schmidt et al. 1997
Slovenia (Kocevje)	m: 156-200 f: 132-222	100% MCP	RT	Huber et al. 1995*
Sweden (Bergslagen)	m: 305±117 f: 97	95% Kernel	RT	Linnell et al. 2001
Sweden (Sarek)	m: 431±83 f: 251±203	95% Kernel	RT	Linnell et al. 2001
Switzerland (North-East)	m: 106±36 f: 74±2	95% Kernel	RT	Ryser et al. 2004
Switzerland (Northwestern Alps)	m: 137 (74-199) f: 76 (45-164)	95% Kernel	RT	Breitenmoser-Würsten et al. 2001

*cited in Breitenmoser & Breitenmoser-Würsten 2008.

Table III–3. Densities of wolves in Europe, calculated / estimated from scientifically robust field method. Comments include additional information, e.g. season, or animals included. Field methods: RT = radio-telemetry; TC = track counts, e.g. snow tracking. Table sorted alphabetically by country.

Area	**Density [ind/100 km²]**	**Comments**	**Field method**	**Source**
Italy (Casentinesi Forests)	4.7 (3.9-5.2)	only pack wolves	TC; direct observations; howling; "saturation census";	Apollonio et al. 2004
Italy (Casentinesi Forests)	5.6	incl. solitary wolves	TC; direct observations; howling; "saturation census"; estimation of solitary wolves	Apollonio et al. 2004
Poland (Biało-wieża P.F.)	0.9-1.5	late winter density, hunted population	TC, RT	Okarma et al. 1998
Poland (Biało-wieża P.F.)	2.0-2.6	late winter density, protected population	TC, RT	Okarma et al. 1998
Poland (Bieszcza-dy)	5.1 (4.3-6.1)	early winter	TC	Śmietana & Wajda 1997
Poland (Bieszcz-ady)	3.3 (3.1-3.4)	late winter	TC	Śmietana & Wajda 1997
Poland (Carpa-thians)	2.1 (1.7-2.5)	Winter density, per map area	TC, howling	Nowak et al. 2008
Poland (Carpa-thians)	1.6 (1.3-1.9)	Winter density, per terrain surface area	TC, howling	Nowak et al. 2008

Table III–4. Territory sizes of wolves in Europe, calculated / estimated from scientifically robust field method. Comments: MCP = 100% minimum convex polygon. Field methods: RT = radio-telemetry; TC = track counts, e.g. snow tracking. Table sorted alphabetically by country.

Area	Territory size [km²]	Comments	Field method	Source
Croatia (Dinarids)	141-160	MCP	RT	Kusak et al. 2005
Germany (Lusatia)	328	MCP	RT	Reinhardt & Kluth in prep.
Italy (Abruzzo)	197	MCP	RT, TC	Ciucci et al. 1997
Italy (Casentinesi Forests)	1.12 (1.06-1.23) packs/100 km²	MCP, summer	TC; direct observations; howling; "saturation census";	Apollonio et al. 2004
Poland (Białowieża P.F.)	173-294	MCP	TC, RT	Okarma et al. 1998
Poland (Białowieża P.F.)	219 (137-323)	MCP	TC, RT	Jędrzejewski et al. 2007
Poland (Bieszczady)	85 (82-90)	based on travel routes	TC	Śmietana & Wajda 1997
Poland (Bieszczady)	85 (82-90)	based on travel routes	TC	Śmietana & Wajda 1997
Poland (Carpathians)	120 (74-172)	MCP, map area	TC, howling	Nowak et al. 2008
Poland (Carpathians)	158 (98-227)	MCP, terrain surface area	TC, howling	Nowak et al. 2008
Slovakia (Carpathians)	146-191	MCP	RT	Findo & Chovancova 2004
Slovenia	403 (259-560)	MCP	RT	Mulej et al. 2013

Appendix IV. Summary: Operationalising Favourable Conservation Status for large carnivores

Achieving a Favourable Conservation Status (FCS) is the general goal for species conservation in the frame of the EU Habitats Directive. The problem of the concept lies in its operationalisation for species as diverse as lichens and lynx. An attempt at this has been made in the "Guidelines for Population Level Management Plans for Large Carnivores in Europe" (Linnell et al. 2008). *"The central challenge associated with operationalising FCS is to make a link between the philosophical / political / legal concept of FCS, the biological concepts of population viability, other existing forms of categorising species status (e.g. IUCN red lists), and the specific distribution patterns and biology of the large carnivores"* (Linnell et al. 2008). The following summary bases mainly on Chapter 5 of Linnell et al. (2008), complemented with some points relevant for wolf or lynx in the Alps.

The concepts of population viability

Demographic viability calculates the probability of extinction for a population of a given size within a specified number of years. Many mathematical models exist and have been tested with empirical data, but there are no agreed-upon standards concerning the best models to use, probability thresholds and time horizons to be considered for "viability", apart from those included in the IUCN's Red List guidelines[1]. But also the latter give enough leeway regarding model details which can influence the outcome. In addition, there is an argument against the usage of population viability analysis (PVA) to set absolute goals for conservation. One of the dangers of PVAs is to make predictions that go too far into the future. Another risk is that populations with an unfavourable PVA may be rashly given up. Nevertheless, many conservation biologists regard PVAs as a most useful tool to compare the relative effects of various scenarios or conservation measures. Populations should be subject to continued monitoring, enabling adaptive management according to the gathered data, so that, if e.g. a flawed PVA resulted in a poor estimate of the Minimum Viable Population (MVP), management can be adjusted in time.

Genetic viability concerns the long term persistence of genetic variation and evolutionary potential, and the avoidance of genetic impoverishment through inbreeding. To assess the genetic viability, it is necessary to distinguish (1) the actual population size N_c from (2) the "number of mature individuals" MI (as required in the IUCN Red List assessment[2]), and (3) the effective population size N_e. N_c is the total number of individuals in the population (e.g. from a census at a given point in time); MI is the number of individuals, which can potentially reproduce, so all animals in the population that have reached sexual maturity; and N_e is the number of individuals (both sexes) that contribute genetic variation to the population size[3]. (In an inbred population, N_e will not only be much lower than N_c –

[1] Depending on the available data different models can be used, e.g. occupancy models, a scalar (unstructured) model, a structured model, an individual-based model or a spatially explicit model. If the model results in a probability of extinction ≥50% in 10 years or 3 generations (whichever is longer; 100 years max.) the population is considered as Critically Endangered CR; a probability of extinction ≥20% in 20 years of 5 generations (whichever is longer; 100 years max.) defines a population as Endangered EN; and a probability of extinction ≥10% in 100 years makes a population Vulnerable VU (IUCN Standards and Petitions Subcommittee 2014).

[2] The definition of MI and the way to count "mature individuals" or "adult individuals" has been a matter of debate for some time regarding the IUCN Red List assessment. The *Guidelines for Using the IUCN Red List Categories and Criteria* (Version 11, IUCN Standards and Petitions Subcommittee, February 2014) clarify that all sexually mature individuals – regardless whether they actually reproduce or not – need to be considered. Hence, in our case, the MI would be the sum of all wolves and lynx ≥2 years living in the Alps.

[3] A more correct definition of $N_{e\ genetic}$ would be: "The effective population size is the size of an ideal population (i.e. one that meets all the Hardy-Weinberg assumptions) that would lose heterozygosity at a rate equal to that

which is normal –, but also considerably lower than MI.) Due to the lack of good empirical data, it is often referred to the 50/500 rule of thumb, referring to the effective population size needed in the short/long term to avoid loss of genetic variation and inbreeding. (Sometimes, the 50/500 is also referred to the population size needed for demographic/genetic viability.) However, this rule of thumb is based on few data, mainly from livestock and fruit flies, and some wildlife experts state that the values should be an order of magnitude larger. Additionally, the relationship between actual and effective population size is very complex and has been estimated for only very few large mammal populations. The effective population size N_e can be expected to be about 10–20% of the actual population size N_c, depending on the one hand on the definition of N_c (e.g. the season of the census in a population with high reproductive output and high juvenile mortality) and on the other hand on the degree of inbreeding, which will reduce N_e compared to N_c and MI. Despite these uncertainties, the important conclusion is that it usually takes a far larger population (e.g. by a factor 10) to maintain genetic viability than for demographic viability.

Inbreeding can considerably reduce $N_{e\ genetic}$ compared to MI or N_c. For instance, the inbred lynx populations in the Dinaric range and the north-western Alps have a strongly reduced N_e if inbreeding is considered. During the International Workshop "Genetic status and conservation management of reintroduced and small autochthonous Eurasian lynx Lynx lynx populations in Europe" held in November 2011 in Saanen, Switzerland, the experts estimated $N_{e\ genetic}$ for the Dinaric population to be about 11 and for the NW Alps about 18 (in a population with a MI of about 54 lynx). In the latter case, Ne would be only 0.33 of the number of independent lynx as estimated from camera trapping (Ch. Breitenmoser-Würsten, pers. comm.). In both reintroduced populations, the inbreeding factor F_{it} is close to 0.25, indicating that all members of the population are as closely related to each other as siblings.

Ecological viability refers to the interaction between a species and its environment. This encompasses both the needs, but also the effects of large carnivores regarding their environment. In recent years, this subject has received much focus in North America, and it was concluded that maintaining ecological viability requires far larger numbers of animals than a simple minimum viable population. In the case of Europe and with regard to wolf and lynx, "ecological viability" could indicate that these species should be allowed to coexist in all habitats suitable for both, the predators and their main prey species. Considering the high adaptability of large mammals – both carnivores and herbivores – such a definition would include a large number of habitats including anthropogenic altered landscapes.

In general, demographic and ecological viability are assessed at the population level, and genetic viability at the metapopulation level. However, populations or metapopulations for wolf and lynx are not easy to distinguish in Europe. Whereas lynx populations as defined by the LCIE (e.g. Kaczensky et al. 2013a) are both demographically and genetically rather distinct (so far), all wolf populations in (continental) Europe are genetically part of the same metapopulation (or will be so soon), and neighbouring populations are also demographically connected.

of the observed population" (D. L. Hartl. 2000. A primer of population genetics. 3rd edition, Sinauer Ass. Sunderland, MA), but the problem of such a correct definition – and with N_e in general – is that it is very difficult to evaluate in a real situation with most often limited information.

Linking the concepts of Favourable Conservation Status and Viability

The definition in article 1 of the Habitats Directive says:

"The conservation status will be taken as 'favourable' when:

- *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 habitat, 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 population on a long-term basis"* (Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora).

The FCS is based on two major Favourable Reference Values (FRV) – the Favourable Reference Range (FRR) and the Favourable Reference Population (FRP) – according to the DocHab-04-003/03 rev 3 and the guidance documents. They are explained as follows:

- 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 species; favourable reference values must be at least the range when the Directive came into force [...]; best expert judgement may be used to define it in the absence of other data"* (Linnell et al. 2008).
- 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 [...]; best expert judgement may be used to define it in absence of other data"* (Linnell et al. 2008).

This being a directive text, like any legal text, it is not based on scientific definitions, which poses some major challenges for the operationalisation of the concept.

The guidance document "Assessment, monitoring and reporting under article 17 of the Habitats Directive" indirectly states that a population must be at least bigger than a MVP to be able to reach FCS. The upper limit is defined by what the potential habitat can support at an "optimum density".

An operational proposal to define Favourable Reference Population

The Favourable Reference Population (FRP) needs to be at least as great as the Minimum Viable Population (MVP). One of the most widespread practical reference definitions for MVP is the IUCN Red List criterion E, which allows a maximum probability for extinction of 10% for a population not to be considered as threatened[4]. This would correspond to the IUCN category of "Near Threatened NT" which is not formally a threat category and therefore a robust benchmark for a minimum population size[5]. However, the necessary Population Viability Analyses (PVAs) require a vast amount of data from years, or even decades, of expensive and invasive fieldwork. As a consequence, only few have actually been performed. Thus, the IUCN Red List criterion D is often used as an alternative.

The IUCN Red List criterion D is based on the estimation of the number of individuals in a population. The threat category "Near Threatened" is reached with a population of 1,000 or more <u>mature</u> individuals in the population. If the assessment is performed regionally, the same benchmark has to be

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[4] The IUCN Red List threatened categories are Vulnerable VU, Endangered EN, and Critically Endangered CR.

[5] NB: Many conservation biologists argue that the benchmark of 10% in the IUCN guidelines is too liberal and instead recommend adapting a benchmark extinction risk of a maximum of 5%.

used. However, if the considered regional population is connected to a neighbouring population to such an extent that immigration can have a significant demographic effect on the extinction probability of the population and the sum of the populations reaches the benchmark, then the threat category for the regional population can be downgraded by one level; i.e. if two connected neighbouring populations exceed the benchmark of 1,000 mature individuals, the regional subpopulation is still considered as not threatened if it exceeds the next lower benchmark of 250 mature individuals (which would classify as "Vulnerable VU" in an un-connected population).

Whether populations can be regarded as connected, of course, always depends on the species concerned, its actual distribution and its dispersal ecology. However, in special cases the connection can also be established and maintained by translocations[6]. This is suggested as an acceptable form of connectivity as long as it is formally included in a management plan at a level that is sufficient for its purpose.

However, such a MVP is truly an absolute minimum population size that can be tolerated as preliminary level for FRP. According to the Habitat Directive guidance documents MVP is only *"a proxy for the lowest tolerable population size"* that can be considered. The reason is that the PVAs resulting in the MVPs are most often based on demographic considerations. Only few include genetic information, the possibility of catastrophic events such as outbreaks of diseases, or the direction and magnitude of changes in the environmental conditions, e.g. climate change. It is therefore recommended that FRP should be defined at significantly higher levels than the MVPs predicted by PVAs.

"*In summary, we suggest that favourable reference population be defined as the sum of the following criteria:*

(1) *The population must be at least as large as when the Habitats Directive came into effect, and,*
(2) *The population must be at least as large (and preferably much larger) as a MVP, as defined by the IUCN criterion E (extinction risk based on a quantitative PVA with <10% extinction risk in 100 years), or criterion D (number of mature individuals).*
(3) *The population's status is constantly monitored using robust methodology"* (Linnell et al. 2008).

An operational proposal to define Favourable Reference Range

Put simply, the Favourable Reference Range (FRR) is the area needed to contain the Favourable Reference Population. However, there are three issues that warrant consideration.

a) Habitat Quality: For example, transport infrastructure can be a source of mortality as well as a barrier to the movement of individuals. The suitability of an area should be assessed before it is included in the FRR.
b) Density: Besides the ecological carrying capacity, which is mainly defined by the habitat quality and prey density, there is also the societal carrying capacity, referring to the willingness of the local community to accept large carnivores in their surroundings and to pay the societal costs of their presence (e.g. damage to livestock, competition for game, fear). These influ-

[6] The artificial fusion of several small and isolated neighbouring populations is called a "managed metapopulation". To assure the genetic viability of each population, some individuals are translocated between the populations, simulating natural dispersal. This is a model primarily for fenced populations (e.g. in fenced PAs in Africa or Asia), may however also be a necessity for wildlife populations in Europe, if increasing landscape fragmentation does not allow for a sufficient genetic exchange.

ence the density of large carnivores in an area, and hence the abundance and the status ("viability level") in this region.

c) Connectivity: As a rule of thumb, one genetically effective migrant per generation poses the minimum amount of connectivity needed between two populations to prevent inbreeding. However, higher rates of migration are needed to result in a significant demographic effect. Linked populations show higher long term viability.

"As a result we generally recommend that Favourable Reference Range be considered larger than the area strictly necessary to support the Favourable Reference Population, and that it attempts to ensure (1) the continuity of distribution within a given population, and (2) the possibility for connectivity between populations" (Linnell et al. 2008).

If a natural connectivity cannot be achieved, artificial connectivity through translocations could be a potentially valuable conservation tool.

An operational definition for favourable conservation status for large carnivores

"We […] suggest that a population can be regarded as having reached FCS if it satisfies all of the following criteria;

(1) '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 habitat' *(Article 1 (i)). We interpret this as implying that monitoring data indicate the population has a stable or increasing trend. We believe that a slight reduction in population size may be permitted if it is a result of response to changes in prey density or habitat quality that are not the cause of direct human action, unless conditions for derogations apply […]. All segments of a population should have stable or positive trends, and not just the population as a whole. And,*

(2) 'The natural range of the species is neither being reduced nor is likely to be reduced for the foreseeable future' *(Article 1 (i)). We interpret this as implying that the overall distribution of the population is stable or increasing. And,*

(3) 'There is, and will probably continue to be, a sufficiently large habitat to maintain its population on a long-term basis' *(Article 1 (i)).We interpret this to imply that the quality and continuity of habitat should be sufficient, and have a stable or increasing trend. And,*

(4) The population size and range are equal to or greater than when the Directive came into force. And,

(5) The Favourable Reference Population size has been reached. According to our proposal this will be set at levels greater than those regarded as being viable using the IUCN Red List criteria E or D. And,

(6) The Favourable Reference Range has been occupied. And,

(7) Connectivity within and between populations (at least one genetically effective migrant per generation) is being maintained or enhanced. And,

(8) 'Member States shall undertake surveillance of the conservation status of the natural habitats and species referred to in Article 2 with particular regard to priority natural habitat types and priority species' *(Article 11) and* 'Member States shall establish a system to monitor the incidental capture and killing of the animals species listen in Annex IV (a)' *(Article 12.4). These statements combine to indicate that the population should be subject to a robust monitoring program.*

Criteria 1-3 and 8 are taken from the text of the Directive, criteria 4 and 6 are taken from the guidance documents, while criteria 5 and 7 are based on our own recommendations" (Linnell et al. 2008).

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Ideally, from a conservation point of view, the metapopulation would consist of connected populations, *each of which is at a level exceeding the minimum threshold for Favourable Conservation Status*[7] (Linnell et al. 2008). It is stated in the guidance documents that **FCS is a positive goal, where the goal should be to make species status as favourable as possible, and not just to have passed a minimum benchmark.** It is also stated in the guidance documents that the FRR can be less than the maximum potential range for wide-ranging species; i.e. not all the historical range has to be recolonised. This gives countries the possibility in some cases to place limits on potential recovery, as well as use derogations to use lethal control in some circumstances, if conflicts are large and difficult to mitigate.

"[T]he absolute minimum requirements that Member States must meet are:

(1) Countries sharing one population, or segments of a population, contribute to ensuring between them that the population reaches and maintains FCS, and
(2) They allow for connectivity between neighbouring populations and segments within the same population, and
(3) Management activities do not create a sink that can influence the FCS of a population of any of its segments, and
(4) Populations should in general not be allowed to go below the level they had when the Directive came into force on their territory" (Linnell et al. 2008).

Finally, natural expansion and recolonisation should be preferred over active reintroduction. Also, the reintroduction within the historic range of the species should not be seen as a Community obligation under the Habitats Directive. However, in the case of small, isolated populations, the translocation of individuals to support it, may actually be necessary to ensure reaching FCS.

[7] This is indeed an ideal view. For some central and western European populations, such as the Jura, the Vosges, etc. even the Category VU under Criterion D (250 mature individuals) is not realistic, as the total extent of habitat is not large enough.

Appendix V. Template for the comparison of national wolf management plans. Adapted from Kaeser & Zimmermann (2012) and supplemented with further categories. The information for Switzerland is filled in as an example (from Kaeser & Zimmermann 2012 with amendments).

	France	**Italy**	**Switzerland**	**Germany**	**Austria**	**Slovenia**
Management plan	MEEDDAT & MAP 2008, Group nat. loup 2013	Genovesi 2002, partly implemented	BAFU 2008, BAFU 2010	No national management plan	KOST 2012, Schäfer 2012	Majić Skrbinšek et al. 2011
Authors of the plan (state, scientists, NGOs)			State: FOEN			
Participation of stakeholders in the development of the plan			- Working Group Large Carnivores (“AG Grossraubtiere”) - Consultation process			
Main goal of the management plan			Focus on conflict reduction			
Population approach/ population aims			No measurable parameters, no population goal/approach			
Operative instruments (compensation, translocations, removals)			Compensation Subventions for prevention measures Removal of stock raiders Regulation			
Evaluative instruments (monitoring)			Genetic analysis Mortalities Damage statistics (Observations)			
Responsibilities state authority			Promotion damage prevention Compensation Definition regulation criteria			
Responsibilities regional authorities			Compensation Enforcement regulation			
Delegated tasks			Livestock damage prevention (AGRIDEA) Monitoring (KORA)			

Appendix VI. Summary of available habitat models of lynx and wolf in the Alps

Model results have always to be interpreted with care. Ecological systems are dynamic and cannot be predicted exactly (Chapron et al. 2012). Therefore, models are always a simplification of the reality and a single model can never precisely predict every aspect of nature's complexity (Zimmermann 2004). Models are always static, retrospective and probabilistic. Their reliability can only be established in the future (Herrmann 2011). Ideally, a model should have the following properties: generality, reality and precision (Zimmermann 2004).

Habitat models can be used to test assumptions, to help decide which data should be collected and to provide information about expected or potential developments. Habitat models link the data, e.g. distribution of a certain species or population, to environmental variables and enhance the knowledge of dynamics (Zimmermann 2004, Marucco 2009). Therefore, habitat models can be helpful in conservation management (Marucco 2009, Herrmann 2011).

The landscape context of suitable habitat patches and the dispersal process of a species are important factors when building habitat models (Jędrzejewski et al. 2008). Thus, to create reliable models to predict species distribution or connectivity, high quality data and information, especially about the habitat use and preferences, and dispersal behaviour of the species as well as the landscape structure, is essential (Kramer-Schadt et al. 2004, Zimmermann 2004). Wolves for example have a complex social system. Therefore, when modelling wolf distribution, connectivity or suitable habitat, it is important to account for differences between pack establishments and solitary or dispersing animals, as the spatial requirements of them differ (Marucco 2009, Marucco 2011). Such differences were not considered in most of the wolf habitat models apart from the ones developed by Marucco (2009, 2011).

It is difficult to set an adequate cut-off value above which the habitat is suitable, especially in the Alps, where large patches of suitable habitat are connected through small strips (Zimmermann 2004). In nature, the probability that similar patches of highly suitable habitat are colonized by a species may change highly due to different connectivity to other (source) patches or due to the species' social system (Jędrzejewski et al. 2008).

Every model type has its advantages and disadvantages. The results of every model should be interpreted with caution and in the context in which it has been built. Model results strongly depend on assumptions made, the quality of variables and parameters, the data quality and quantity, the chosen threshold suitability values, the choice of background locations and the methods applied (Becker 2013, Chapron et al, no date). Moreover, models are sensitive to the data origin used to calibrate the model (Zimmermann & Breitenmoser 2007). Models which use data collected in another area than where the habitat is modelled or models which use data only from a part of the entire study area and where then the results are extrapolated to the whole study area have to account for different environmental and perhaps even social factors. Care has to be taken as extrapolation often creates problems as habitat characteristics can differ leading to a change in the model coefficients (Zimmermann 2004). Thus, an important question to be asked is in what extent available information of an area can be transferred to others or not and if yes what has to be considered when doing so (Schadt 2002). Additionally, it is important to account for uncertainties and for possible individual detection heterogeneity when modelling species habitat or distribution.

The ignorance of detection heterogeneity can lead to biases, especially to underestimations (Cubaynes et al. 2010, Caniglia et al. 2012, Chapron, no date,).

As mentioned above the distribution and the quality of the used data and the selection of input parameters are important points. In the model of Herrmann (2011), for example, the predicted habitat suitability for wolves changed noticeably depending on the data used (wolf monitoring data from France or chance observations from Switzerland). As wolves are more likely to be observed by humans close to roads or highways, the model, based on chance observations, showed a correlation between these two variables. Consequently, this wolf suitable habitat model was considered to not represent typical wolf behaviour but rather display a biased image (Herrmann 2011). Moreover, the chance observations from Switzerland were only from lonely male wolves which have a different behaviour than wolf packs (Herrmann 2011).

Depending on the input, applied model techniques and made assumptions, the model output can highly differ (Fechter & Storch 2014). Using Germany as an example, Fechter & Storch (2014) showed that, depending on the different model input parameter sets, the area of potential suitable wolf habitat varied greatly, over 800% in some cases.

The inclusion of critical habitat features and biotic variables such as prey species, predation or competing species has been shown to increase model performance and not incorporating them can lead to prediction errors (Zimmermann 2004, Becker 2013). The distribution of ungulates, main prey of wolf and lynx, is strongly correlated to habitat and land cover. Therefore, it is assumed that adding a prey layer could lead to an over fitting of the model (Doswald et al. 2007). The presence of prey was not or could not be included in all models (Table VI-1, Table VI-2).

The factor human impact, thought to be one of the main factors affecting wolf and lynx mortality and distribution, is not a simple variable and its impact is very difficult to evaluate and its future development is not known (Zimmermann 2004). It cannot be mapped easily and only be modelled indirectly by taking into account other variables such as road and railway density, human population density and land use (Zimmermann 2004).

Not all variables indicated by a model to be important predictors for the species presence or absence must be of biological importance for the species. The general linear model of Zimmermann & Breitenmoser (2002) for example, showed slope and elevation as the most powerful variables predicting lynx occurrence. However, this is not typical for lynx but was for the study area where forested areas were correlated with elevation and slope as a result of human activities (Zimmermann & Breitenmoser 2002, Zimmermann 2004).

Logistic regression models based on presence-absence data are useful in regard to describe the relationship between species habitat requirements and the environmental conditions and they can be used to predict the amount of potential habitat (Schadt 2002). However, logistic regression models, as used by Massolo & Meriggi (1998), Glenz et al. (2001), Schadt (2002) or Signer (2010), do not consider false negatives (e.g. non-detection does not necessarily imply absence), which can lead to biased results when false negatives are habitat dependent (e.g. if in a certain habitat type, the species has a lower chance to be detected) (Marucco 2009, 2011). However, this is also influenced by the resolution applied in the model. Regression models do not include any temporal dynamics or the species' social system and look statically at the relationship between a species and its environment (Marucco 2009). Care has also to be taken in regard to the interpretation of logistic regression model results as they may include landscape variables which are not connected to the species requirements

(Schadt 2002). Another problem concerning logistic regression is dealing with spatial autocorrelation of the dependent variable which can lead to over-parameterised models (Schadt 2002).

Contrarily to logistic regression models, occupancy approaches, when applied to large-scale surveys, account for false positives by including detection probability (Marucco 2009). More complex occupancy models, so called multi-seasonal occupancy models as applied by Marucco (2009, 2011) consider also temporal dynamics but they need a lot of high quality data and robust sampling.

Spatially explicit individual-based models SE-IBMs as applied by Schadt (2002), Kramer-Schadt et al. (2005), Marucco & McIntire (2010) seem promising approaches (Zimmermann 2004). These models imply the ability to estimate values for unknown parameters, to develop explicit hypotheses and to order existing knowledge (Zimmermann 2004). SE-IBM allow accounting for internal complexity within a population by linking individual traits to social system characteristics, thus, including spatial, social and behavioural factors and trying to consider the different aspects of live of the animals modelled (Marucco 2009, Chapron et al. 2012). SE-IBMs follow the fate of each individual in a population and consequently include all aspects influencing the individual behaviour (Chapron et al. 2012). SE-IBMs can include different mortality scenarios, model spatial reintroduction situations and cope with heterogeneity in space. Therefore, SE-IBMs can deal with habitat fragmentation (Chapron et al. 2012). Applying individual-based models is considered as a relevant choice especially when modelling wolf population dynamics (Chapron et al, no date). However, multi-seasonal and individual-based models need a lot of high quality data and robust sampling design which are constraints and disadvantages of these models (Marucco 2009). SE-IBMs are especially sensitive to errors made in estimating survival and dispersal rates (which may be strongly influenced by the landscape matrix disperser encounter) (Zimmermann 2004). Results of the SE-IBM are mainly a consequence of the model structure and assumptions. Consequently, a poorly structured or incomplete model will produce only incomplete results. For example when, modelling lynx habitat or distribution results can lead to false interpretations if for example effects of linear barriers are not included (Zimmermann 2004).

Ecological Niche Factor Analysis ENFA uses presence data only. It is advised to be used when absence data are not available, unreliable or meaningless. ENFA compares the species' environmental niche to the environmental characteristics of the entire study area to build the habitat suitability model (Zimmermann 2004).

It is assumed that using different models and pooling together their results should generate a more robust confidence (Chapron et al, no date). To include simpler, more general models and complex ones into the analyses is thought to be important (Chapron et al, no date). Of course, the assumption that the combination of different models enhances their reliability can be tested only in the future.

Table VI–1. Lynx models: Summary of considered lynx habitat models in the Alpine countries or parts of the Alpine Arc.

Paper	Model Area	Model & Method	Data origin	Variables (combined)	Results
Alpine range					
Becker 2013	Alpine range (as defined by Alpine Convention) (~190,000 km^2)	Maxent model Only presence data used	Data on lynx presence from Switzerland, Italy, Austria and France (GPS and VHF data)	• Road density • Distance to settlement • Human influence index • Elevation • Slope • Vector ruggedness measure • Land cover data	• Mean temperature of coldest quarter, precipitation of driest month, CORINE, Elevation & slope were important variables. • High habitat suitability across the Alps, although habitat is quite fragmented with many barriers. • Approx. 103, 600 km^2 (54%) are predicted as suitable area. • Areas seem to be reasonably well connected.
		Threshold application of 0.3323 and 0.4609 on Maxent habitat suitability model and inclusion of major barriers (highways, rivers, high elevation areas			• 32 different patches identified with sizes ranging from 57 to 17,376 km^2. • 22 patches greater than 400 km^2, considered large enough to support individual lynx subpopulations. • Patches are divided by major barriers resulting in a fragmented landscape. • Estimated population for the Alpine range: 1035-3107.
Signer 2010	Alps (Alpine Convention) (~190,000 km^2)	Logistic regression from Zimmermann & Breitenmoser (2007) for the Jura adapted for Alps.	Sighting data from Austria, Econnect pilot area Northern Limestone	• Shrub • Forest • Altitude • Declivity	• Eastern Alps are predicted to have more highly suitable lynx habitat than the western or central Alps.
		Morphological Spatial Pattern Analysis Using GUIDOS Graph base approach to look at connectivity			• Eastern Alps are indicated to have larger patches of lynx core areas - lynx core areas in western and central Alps are patchier distributed. • Highways are major barriers to lynx migration • Settlements have little negative impact on lynx.

Paper	Model Area	Model & Method	Data origin	Variables (combined)	Results
Zimmermann 2004, Zimmermann 2003	Alpine range	Ecological Niche Factor Analysis (ENFA) Based on habitat suitability map, cut off value chosen arbitrarily so that 80% of cells were included and threshold set at 70% of presence cells were included and overlaid with barrier map. Presence data only	Radio-telemetry data from Swiss Alps and Jura	• Forest • Shrub and/or herbaceous vegetation • Open spaces • Pastures • Roads • Distance to large and medium towns • Heterogeneous agricultural areas	• Lynx were essentially linked to forest and shrubs and/or herbaceous vegetation. • Lynx avoid areas of heterogeneous agriculture. • Total area of suitable habitat in the Alps is about 93,579 km^2. • 37 suitable habitat patches from 50 to 18,711 km^2. • 16 patches in the Alps were larger than 380 km^2. • Small bands of habitat connecting suitable habitat patches may act as bottlenecks and impede movements of lynx. • Forest and shrubs are known to provide good shelter and food for roe deer and chamois. • Alps could host a population of 961-1827 resident lynx.
France					
Rolland et al. 2011	French Jura Mountains ~16,000 km^2	Model of lynx occurrence based on Mahalanobis distance factor analysis (MADIFA) and site-occupancy modelling	Presence signs of lynx (scats, hair, tracks, visual observation, kills, pictures) in study area	• Land cover • Railway density • Road density • River density • Distance to highway • Distance to railway • Distance to river • Prop. of forest cover	• High road and river density led to a lower habitat suitability • Higher forest proportion indicated higher habitat suitability • Proportion of forest cover was important variable for lynx occupancy • Main factors structuring lynx occurrence: road and river density and proportion of forest cover • Lynx occurrence was lower in periphery than in centre of study area • Negative effect of road density on occurrence
Basille et al. 2008	Vosges Mountains ~16,500 km^2	Ecological Niche Factor Analysis (ENFA)	Lynx presence signs (tracks, hairs, scats, sightings, carcasses, kills) in Vosges Mountains	• Prop. of agricultural areas • Distance to artificial areas • Elevation • Prop. of forest • Distance to highways • Distance to main roads • Distance to railways • Distance to rivers • Density of roads • Slope	• Lynx was searching for high elevation (slopes), dense forest cover & avoided highways & areas of high agricultural use. • Selection for high elevation & steep slopes due to high forest and low proportion of agricultural areas & highways. • Lynx was restricted to areas with low agricultural use, far from highways and with high forest proportion • Weak influence of artificial areas on habitat use • Lynx can support high human activity if there are enough forested area patches • Critical habitat features included proportion of forest and agricultural areas, and distance to highways.

Paper	Model Area	Model & Method	Data origin	Variables (combined)	Results
Switzerland					
Zimmermann 2004	Switzerland	Ecological Niche Factor Analysis (ENFA)	Radio-telemetry data from Swiss Alps and Jura	• Fringe length • Forest • Other wooded areas • Agricultural areas • Urban areas • Roads and railways • Southern aspect • Mean elevation • Mean slope • Mean distance to roads • Mean distance to settlements • Human population density	• Lynx were linked to forest areas with high values of fringe length & high levels of extensive agricultural areas. • Tended to avoid zones of permanent human activities: Presence of lynx was negatively correlated to frequency of intensive agricultural areas and urban areas. • Lynx when ranging in good quality habitats are adapted to human presence. • Lynx were linked to extensive agricultural areas and other wooded areas, to close forest and open forest areas with high values of fringe length. • Distance to road was not indicated as important factor.
Doswald et al 2007	North-western Alps, Switzerland (~2,800 km^2)	Expert model Based on Multiple Criteria Decision Making (MCDM) and Analytical Hierarchy Process (AHP)	Experts (Switzerland, Austria, Italy, Germany & Slovenia, game wardens part of north-western Swiss Alps)		• Land cover and forested areas have been deemed important. • Elevation was also rated quite highly. • Proximity to motorways and towns is thought to be more disruptive to the lynx than the proximity to railways and minor roads.
Zimmermann 2004	Jura Mountains	Probability model based General linear models Cost distance analysis with ArcView	Radio-telemetry data from the Swiss Jura Mountains	• Urban fabric • Industrial areas • Artificial areas • Arable land • Permanent crops • Pastures • Forests • Shrubs • Open space • Wetland and water bodies • Elevation • Slope	• Four possible corridors connect the Jura Mountains to the adjacent ranges. • Jura Mountains are separated from the French Alps by a 7.3 km long corridor passing the Rhone River and a main road (no insurmountable dispersal barriers to lynx). • Two other corridors exist between Jura Mountains and French Alps. One connecting Jura Mountains to the Salève and one the Salève to the Alps. • A corridor connects the Alps and the Chartreuse.

Paper	Model Area	Model & Method	Data origin	Variables (combined)	Results
Zimmermann & Breitenmoser 2002	Swiss Jura Mountains	Generalized linear models to select predictors	Radio-telemetry data from Swiss Jura Mountains, Presence only data	• Human population density • Built-up areas • Roads and railways • Slope • Elevation • Eastness • Northness • Land cover data • Fringe length	• Slope & elevation were most powerful variables predicting lynx presence/absence. Forested area in study region was correlated with elevation & slope as result of human activities. • In the Alps an area of approximately 93,600 km^2 has been predicted as suitable. • Lynx were not located in all favourable lynx zones within the study area because peripheral spots of good habitat might not be connected to the lynx zone and might be occupied by neighbouring lynx or surveillance density might have been insufficient.
Germany					
Schadt et al. 2002a	Germany and large adjacent forest covered areas in neighbouring countries: Bohemian forest, forest along German-Czech border and German-Polish border, Vosges Mountains, without Alps (~374,000 km^2)	Rule-based habitat model Rules based on current knowledge on lynx biology. Patch connectivity model (using cost-path analysis)	Based on radio telemetry data from Swiss Jura Mountains and Poland	• Land use data • Urban areas • Agricultural areas • Pastures • Forest • Open areas with natural vegetation • Water bodies • Land cover data	• 58 patches of suitable habitat in Germany and extending over the border with total area of 54,260 km^2. • Patches of suitable habitat located in the low mountain ranges of south and central Germany, and in large forests in north and east of Germany. • 10 nuclei with a total area of 38,400 km^2. • 10 nuclei could host ~230 resident females and 150 resident male lynx. • Only 5 nuclei with high suitability could host a population size of 111 resident females and 71 male lynx.

Paper	Model Area	Model & Method	Data origin	Variables (combined)	Results
Schadt 2002, Schadt et al. 2002b	Germany 358,000 km^2 Included neighbouring forest areas in Poland, Czech Republic, France, Belgium. Excluded Alps as the habitat requirements are different	Home range suitability model using Logistic regression	Radio-tracking data from Swiss Jura Mountains. Model validated with telemetry data from Czech Republic and Slovenia	• Urban areas • Agricultural land • Pasture • Forest • Non-wooded semi-natural areas • Wetlands • Water bodies • Roads	• Lynx radio locations showed a clear tendency for avoidance of intensively used land-use types and a preference for forest. • Lynx home ranges tended to have more forest cover and a greater perimeter of forest. • About 81% of Germany consisted of unsuitable area. • Area of 29 119 km^2 predicted as suitable for Germany. Suitable areas of habitat mainly concentrated in the low mountain ranges of Germany. • Considering only areas with p>0.5 and a size >100 km^2, an area of 24,119 km^2 of Germany or 32,266 km^2 for Germany and neighbouring forest areas was indicated as suitable. • ~370 lynx predicted for suitable patches in Germany.
Schadt 2002, Kramer-Schadt et al. 2004	Germany and large adjacent forest covered areas in neighbouring countries: Bohemian forest, forest along German-Czech border and German-Polish border, Vosges Mountains (~358,000 km^2)	Spatially explicit individual-based, dispersal model (SEDM) Behavioural rules on dispersal characteristics based on general knowledge of dispersal and on movement analyses	Calibrated with telemetry data from Swiss Jura mountains. Data of long-term field studies from Switzerland, Poland and Spain.	• Mortality risks • Exponent of power function • Max. nbr of intraday movement steps • Prob. of stepping into matrix • Prob. of keeping the previous direction • Max. residence in matrix cells • Mortality probability on highways • Mortality probability on main roads per crossing • Baseline mortality probability per day	• Most important factor for determining lynx movement was the availability of dispersal habitat. • Dispersers used forests significantly more than open space. • Patch connectivity was also time dependent. • Connectivity between patches could be enhanced when enough dispersal habitat is available. • Not only the distribution of dispersal habitat limited patch connectivity but also factors contributing to a high mortality (e.g. traffic system). • Main factor hindering patch connectivity was road mortality • The Alpine part was indicated as suitable habitat and as target patch > 100 km^2.

Paper	Model Area	Model & Method	Data origin	Variables (combined)	Results
Schadt 2002 continued		Spatially-explicit individual based Population simulation model Basis for model is validated habitat model for lynx in Germany (consisting of barriers, matrix, dispersal and breeding habitat). Landscape submodel, population submodel, dispersal submodel	Data from French and Swiss Jura Mountains, Czech Republic and Slovenia		• Source patches were not interconnected except along the German Czech border. • Survival rate of adults with territories was the most sensitive parameter. • Best management strategy for success of reintroduction would be reducing mortality of residents in source patches. • Viable population in the Harz Forest under current landscape situation only possible when mortality rates of resident animals are kept very low. • Viable population of lynx in Germany principally possible but only under the precondition of low mortality for resident and dispersing lynx. • Even short highway sections could be significant barriers.
Kramer-Schadt et al. 2005	Germany 358,000 km^2	Individual-based spatially explicit model to simulate spatiotemporal population dynamics of lynx in Germany Landscape submodel (logistic regression, based on Schadt et al. 2002b, habitat types: breeding, dispersal, matrix & barrier habitat)	Data from Switzerland, Germany, Czech Republic and Slovenia	• Urban areas • Agricultural land • Pasture • Forests • Non-wooded semi-natural areas • Wetlands and water bodies • Proportion of area used extensively by humans • Main roads, motorways • Main rivers	• Population in Harz Forest under the current landscape situation would only be viable if mortality rates of resident animals could be kept very low. • Under expected mortality scenarios the probability of a lynx reintroduction into the Harz Forest succeeding was only about 0.5 for a time window of 50 years. • Viable populations would be possible in other major patches (Thuringian Forest, Black Forest & Palatinate Forest) but only assuming low mortality for resident and dispersing lynx). • 59 patches, 11 considered as source patches: North-Eastern Forest, Lüneburg Heath, Harz Forest, Rothaar Mountains, Erz Mountains, Thuringian Forest, Spessart, Bavarian Forest, Northern Black Forest, Southern Black Forest, Palatinate Forest.

Paper	Model Area	Model & Method	Data origin	Variables (combined)	Results
Kramer-Schadt et al. 2005 (continued)					• If more than five females and males are released, mortality rate of dispersers plays important role for survival of population.
		Individual based dispersal submodel (developed in Kramer-Schadt et al. 2004).		• Correlation factor • Prob. of stepping into matrix • Maximum nbr of intraday steps • Exponent of step distribution • Annual mortality rate of dispersers • Daily mortality rate of dispersers • Mortality rate per crossing event of road/main river	• Survival of residents was most important factor for establishing viable population • Patch connectivity (at least one female settling in other patch in 50y and 100 simulations) occurred for each source patch and mortality scenario. All source patches are interconnected. • Population spread across Germany was very restricted. • Different mortality scenarios affected probability that reintroduction is succeeding.
		Demographic model	Demographic parameters of the model based on published data from Switzerland, Spain & Poland	• Non-overlapping core area size of female home ranges • Males overlapping females • Surviving subadults starting to disperse per reprod. female • Sex ratio of kittens • Reproduction rate • Annual mortality rate of residents	
		Home range selection model		• Nbr of released males and females	

Paper	Model Area	Model & Method	Data origin	Variables (combined)	Results
Herdtfelder 2012	Baden-Württemberg, Germany; Jura Mountains, Switzerland (~35.750 km^2)	Spatially explicit habitat model	Based on telemetry data from Jura Mountains (Switzerland)	• Land use • Elevation • Roads • Slope • Exposition • Sex • Distance to next road • Distance to next settlement	• Predicted certain preference for areas with slope of up to 50 degrees. • Lynx preferred forest, followed by scrub, open areas & urban areas. They also seemed to prefer steep slopes when these are outside forested areas. Slopes seemed to be highly used as day resting sites. • Preference for slopes explained through the occurrence of forest in slopes. • Lynx predicted to avoid the proximity to roads and slopes with north-east exposition. • Highways with very high traffic intensity seemed to act as barriers and animals do not cross. • Traffic accidents in Switzerland depended on 3 factors: habitat suitability in the periphery of 400 m of the road, the category of the road (indicator for the traffic intensity) and the distance to larger settlements.
Austria					
Rüdisser 2001, Rüdisser 2002, Rüdisser 2009, Rüdisser & Martys 2002	Western Austria, Voralberg, Tyrol, western Salzburg (~19,462 km^2)	Expert habitat model Theoretical GIS based habitat model		• Non-forested areas and settlement areas are assumed as unsuitable • Forest areas smaller than 100 km^2 must be no further than 400 m away from the next forested area • Habitat quality positively affected (1) if distance to next settlement or major road is more than 1 km; (2) compact connected forest areas larger than 30 km^2 are within a circle of 5 km.	• An area of 11,356 km^2 (59%) was predicted as suitable lynx habitat. • Estimated a potential population size of 101–247 (0.9–2.2/ 100 km² suitable habitat) for western Austria.

Paper	Model Area	Model & Method	Data origin	Variables (combined)	Results
Slovenia					
Potočnik et al. 2009	Slovenia, Croatia, Bosnia and Herzegovina	Population dynamic model of Dinaric lynx population Population viability analysis (PVA)	Monitoring data Demographic parameters estimated knowledge about lynx biology	• Demographic • Habitat • Environmental	• Adult survival was the most important demographic parameter • Changes in survival rates of subadult & adults and habitat quality with regard to prey availability had a major impact on population growth dynamics • Survival of adult and subadult lynx were directly influenced by human activities (traffic mortality, legal hunting and mainly poaching) • Habitat quality had a significant role in lynx population dynamics and viability – it had a major impact even when there was only a minimal change in adult or subadult survival rate

Table VI–2. Wolf models: Summary of considered wolf habitat models in the Alpine countries or parts of the Alpine Arc. Variables including prey are written in bold.

Paper	Model Area	Model & Method	Data origin	Variables (combined)	Results
Alpine Range					
Herrmann 2011	Alpine range (as defined by Alpine Convention) (~190,000 km^2)	Presence-only model Maxent	Data from breeding packs in the French Provence-Alpes-Côte d'Azur and Rhône-Alpes regions. Chance observations from Switzerland	• Elevation • Land cover data • Ruggedness • Urban area density • Road density	• Predicted 245-320 wolves in F, 370-485 in I, 150-200 in CH, <1 in FL, 35-45 in D, 375-490 in A, 25-35 in SLO and 1200-1580 in the whole Alpine Convention area. • Suitable habitat just below 1,000 m to 3,000 m. • Suitable habitat in km^2 per country: 18,875 in F, 28,520 in I, ~12,000 in CH, 60 in FL, 2,610 in D, 28,880 in A, 1,900 in SLO and 92,870 km^2 in the whole Alpine Convention Area. • Preference for areas with high forest and shrub land density
Marucco 2011	Alpine range (as defined by Alpine Convention) (~190,000 km^2)	Multi-season occupancy model (Marucco 2009) Spatially explicit individual-based model (Marucco & McIntire 2010) Morphological spatial pattern analysis threshold 0.5 and 0.8.	Detection and non-detection data from Western Italian Alps	• Human disturbance (roads, settlements, lakes) • **Presence of red deer** • Forest cover area • Rock cover area	• Human disturbance & rock-area cover had negative, presence of red deer and forested-area cover had positive effect. • Road density, human settlements, low forest cover and high rock elevation were negatively connected to wolf presence. • At 0.5 threshold: 70.41%, 0.8 threshold 25.96% (48,357 km^2) of study area are identified as suitable wolf pack habitat.
Falcucci et al. 2013	Alpine range ~300,000 km^2)	Presence-only model using partitioned Mahalanobis distance ($D^2(k)$)	Presence only data from French and Italian Alps (observations, scats, GPS locations)	• Human population density • Distance to infrastructure • **Prey species richness** • Elevation & Slope • Terrain ruggedness index • Land cover data	• Highest suitable areas located further from main roads and railways and at an average elevation of 1,603 m • Suitable areas contain natural vegetation, low human population density & high prey species richness

Paper	Model Area	Model & Method	Data origin	Variables (combined)	Results
Italy					
Massolo & Meriggi 1996	Northern Apennines (Italy) (3,289 km^2)	One-way ANOVA Discriminant Function Analysis (DFA)	Data from Apennines (Italy) (scats, howling, snow tracking, predation, direct observation)	• Land cover data • Forest cover diversity • Landscape diversity • **Ungulate diversity** • Urban areas, village density • Road length • Hunting pressure • Inhabitant density • **Livestock abundance**	• Human pressure negatively influences wolf presence. • Prey abundance and forest cover made wolf presence more probable.
Massolo & Meriggi 1998		Logistic regression model (dichotomous and polytomous logistic regression)			• North exposure, arable land & hunter density limit wolf presence. • South exposure, red-deer abundance, forest diversity & cover, scrub extension & wild ungulate abundance had positive impact. • Suitable habitats: where wild prey is abundant and diversified, human impact is low & forest cover abundant
Boitani et al. 1998	Italy (~250,000 km^2)	GIS based model Multivariate analysis Linear correlation Mahalanobis distance	Data from Dupré et al. (no date)	• Altitude • **Livestock density** • Road density • **Number of ungulate species** • Shannon diversity index • Land cover data • Dumping site density • Human density • Urban settlement extension	• Only dumping sites & extension of forest showed linear correlation to wolf presence probability, but only forest seems to be a useful index. • Suitable habitat (with probability > 0.5) covered less than 15% (37,265 km^2), very highly suitable habitat covered 3.71% (9,294 km^2). • Large parts of the Alpine range are indicated as highly suitable. • Single variable is not suitable to describe wolf distribution adequately in complex environments.

Paper	Model Area	Model & Method	Data origin	Variables (combined)	Results
Corsi et al. 1999	Italy	Spatially explicit model (based on multivariate analysis of GIS data) Discriminant analysis model	From Italy (presence and absence data, dead wolves)	• Land cover data • Urban settlement • Human density • Dumping site density • **Sheep density** • Shannon diversity index • Shannon dominance index • **Number of ungulate species**	• Human pressure & attitude are probably most important factors impacting wolf distribution. • About 2,900 km^2 of wolf core habitat predicted in Italian Alps, mainly in the eastern part.
Marucco 2009	Western Alps of Piemonte Region Italy (~25,388 km^2 of which 11,334 km^2 in the Alps)	Multi-season occupancy model (includes temporal dynamics, social structure)	Data from Western Alps (scats, tracks)	• Land cover data • Elevation • Road density • Settlements • **Ungulate presence**	• Presence of red deer & forested area cover positively and human disturbance & rock-area cover negatively influence wolf occupancy. • Wolves still have a quite large amount of suitable habitat to recolonize (especially in Eastern part). • Human disturbance highly negatively influences Wolf presence. • Wolves in the Alps primarily avoid people & rocky areas, then occupy remaining forest habitat and pasture areas (especially where red deer are).
Marucco & McIntire 2010	Western, central and eastern Italian Alps	Spatially explicit individual-based model based on Wolf habitat suitability model (Marucco 2009)			• Wolf population in the Alps is supposed to increase mainly from West to the East • Number of packs and population size predicted to increase from 2008 through the next 15 years on Italian Alps range. • Ligurian-Maritime Alps main source for dispersing wolves repopulating the Alps from 1999 to 2008. • Source of dispersing wolves will shift to Cozie Alps (i.e. NW Alps) until 2018. • By 2013 source continue to shift eastward to Graie Alps, west of the lakes to Switzerland, allowing recolonisation of Eastern Alps.

Paper	Model Area	Model & Method	Data origin	Variables (combined)	Results
Germany					
Fechter & Storch 2014	Germany	Spatially-explicit, predictive rule-based model	Data from Germany (Lausitz) (tracking, camera traps, observations)	• Road density • Human population density • Wolf home range size • Core areas of wolves • Land cover data	• Road density and human population density were good indicators for habitat suitability. • Most suitable habitat indicated in east and north-east Germany. • Bavarian Alps are predicted as highly suitable. • Priori assumptions and selection of different input data and techniques influence highly the model results
Switzerland					
Landry 1996; Landry 1997b	Valais, Ticino, Grisons, Switzerland	GIS based model based on Dupré et al. (year)		• Land cover data • Elevation • Human density • Urban area	• In summer, wolf was predicted to occupy 79.6% (15,142 km^2) of the area. Forested areas and alpine meadows are important. • Wolves are predicted mostly in the mountains. • In winter, only 32.25% are predicted to be suitable for wolves. • Number of wolves able to live in a certain area depends on available territory in winter. • Wolf presence depends on human pressure
Glenz 1999; Glenz et al 2001	Valais, Switzerland (5,224 km^2)	Stochastic model based on logistic regression of Massolo and Meriggi (1996)	Data from northern Apennines (Italy)	• Land cover data • Population density • Urban areas • NW-exposition • Minimum altitude • **Wild ungulate diversity index**	• Population density, arable lands & ungulate diversity index had highest effect. • 19% (1,142 km^2) were predicted as suitable habitat (p>50%) • 606 km^2 suitable habitat with probability over 75%. • 260 km^2 were predicted as suitable reproduction area. • Suitable habitat is mainly forest areas along the main valley and smaller side valleys. • Under 800 – 900 and over 1800-2000 m, habitat is unsuitable for wolf • Wild ungulate diversity index was the habitat variable with the highest influence.