Defining and applying the concept of Favourable Reference Values for species and habitats under the EU Birds and Habitats Directives

Examples of setting favourable reference values

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The official guidelines on reporting under Article 17 of the Habitats Directive, including on setting Favourable Reference Values, are available from the Reporting Reference Portal (http://cdr.eionet.europa.eu/help/habitats_art17). They were adopted by the Habitats Committee and endorsed by the European Commission (DG Environment). Therefore, in case of conflicting definitions, approach or examples, the above-mentioned official guidelines take precedence.

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Photo cover: Habitat type 6510 with *Fritillaria meleagris*, the Netherlands (John Janssen)

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Preface

The Birds and Habitats Directive ensure the conservation of a wide range of rare, threatened or endemic animal and plant species as well as characteristic habitat types in Europe. In reporting both directives use 'distance to target' measures regarding conservation status. The Habitats Directive considers explicit favourable reference values while the Birds Directive requires to maintain bird populations at a level which corresponds to their ecological, scientific and cultural requirements.

This report presents examples of setting favourable reference values following the methodology of the accompanying Technical report and in agreement with the Explanatory Notes and Guidelines for reporting under Article 17 of the Habitats Directive for the period 2013–2018 (http://cdr.eionet.europa.eu/help/habitats_art17).

The study was commissioned by the EC under the service contract 'Defining and applying the concept of favourable reference values for species and habitats under the EU Birds and Habitats Directives' (Service contract No. 07.0202/2015/715107/SER/ENV.B.3).

The work was followed and reviewed by the Ad hoc group on 'favourable reference values' of the Expert Group on Reporting under the Nature Directives, and supported the general objective of the ad hoc group (chaired by EEA), namely to improve the guidance related to the setting and reporting of favourable reference values under the nature reporting, and contribute to further harmonise approaches between Member States.

1 Cetaceans

Peter G.H. Evans

1.1 Common bottlenose dolphin (*Tursiops truncatus*) in the European Atlantic

Step 1.1 - Biology of the species
The common bottlenose dolphin (*Tursiops truncatus*) is one of the most widely distributed members of the family Delphinidae, which comprises 38 species of dolphins including larger species such as long-finned pilot whale and killer whale. Bottlenose dolphins are social animals, forming groups that at times may number in the tens or hundreds, although normally they live in smaller sub-groups, often referred to as a fission-fusion society with individuals associating with one another for varying lengths of time. Individuals commonly cooperate with one another in activities including babysitting and coordinated hunting. Adult females and young preferentially associate with other female-young pairs whilst in some regions adult males have been reported forming stable alliances with other males. Males reach sexual maturity at around 10-15 years of age, and females rather earlier, at 5-13 years. Age at sexual maturity can vary between regions and within a region, between individuals. Males may reach >20 years before attaining actual breeding status. On reaching sexual maturity, females typically give birth to a single calf every three years although this too can vary from 2-8 years even within the same population. Males may live to 40-45 years and females to c. 50 years. Generation length has been estimated at 20.6 years (21.1 years in populations at a stable state). Bottlenose dolphins have catholic diets including pelagic, demersal or benthic fish, although tending to reflect the habitat in which groups are living. More detailed accounts of the biology of the species can be found in Wells *et al.* (1987), Wells & Scott (1989), Connor *et al.* (2000), Reynolds *et al.* (2000), and Wells & Scott (2009).

Step 1.2 - Spatial scale of functioning
The common bottlenose dolphin has a worldwide distribution in tropical and temperate seas of both hemispheres. In the North Atlantic, it occurs from Nova Scotia in the west and the Faroe Islands in the east (occasionally as far north as northern Norway and Svalbard), southwards to the Equator. The species also occurs throughout the Mediterranean and in the Black Sea. It is largely absent from the eastern North Sea, Inner Danish Waters and the Baltic (Reid *et al.* 2003).

In many parts of the world, a coastal bottlenose dolphin ecotype has been distinguished from an offshore one, each having different ecologies, food preferences and movement patterns (Walker 1981, Mead & Potter 1995, Curry & Smith 1998, Hoelzel *et al.* 1998, Perrin *et al.* 2011). Within these ecotypes there may be further population sub-structuring leading to separate stocks or management units. The definition of a management unit is a demographically independent biological population (Palsbøll *et al.* 2007, Evans & Teilmann 2009, NOAA 2016). It is usually based upon a variety of lines of evidence including genetics, morphometrics, distribution and movements (i.e. connectivity), and life history parameters.

While high mobility of the species facilitates interaction and gene flow over large distances (Hoelzel 1998, Querouil *et al.* 2007), bottlenose dolphins can also display fine-scale genetic population structure resulting from localised adaptations over small spatial scales (Ansmann *et al.* 2012). Genetic differentiation between neighbouring populations regularly occurs and may be related to habitat borders (Natoli *et al.* 2005, Bilgmann *et al.* 2007, Wiszniewski *et al.*, 2009), sex-biased linked dispersal (Möller *et al.*, 2004; Bilgmann *et al.*, 2007; Wiszniewski *et al.* 2010), niche specialisation (Louis *et al.* 2014a), anthropogenic activities (Chilvers & Corkeron 2001), and through isolation by distance without apparent boundaries separating populations (Krützen *et al.* 2004, Rosel *et al.* 2009).

Consequently, bottlenose dolphins tend to be subdivided into small discrete coastal populations residing relatively close to shore and a much larger wide-ranging offshore population. The relationships both within and between those coastal and offshore populations often remain unclear (Rosel *et al.* 2009, Toth *et al.* 2012, Richards *et al.* 2013, Louis *et al.* 2014b).

In the Northwest Atlantic, sixteen separate coastal stocks or management units and one offshore one have been recognized (Waring *et al.* 2016). In Atlantic European waters (MATL biogeographic region, reported under HD Art 17 by ES, FR, IE, PT and UK), bottlenose dolphins consist of a large and wide-ranging offshore population and much smaller coastal populations The preferential habitat for the offshore ecotype is over the outer continental shelf and shelf break (Reid *et al.* 2003, Certain *et al.* 2008). The coastal populations are fairly resident and tend to inhabit smaller areas close to shore. Ten coastal assessment (= management) units for bottlenose dolphins have been identified, largely from photo-ID studies (ICES 2013, 2016). These coastal populations are also potentially exposed to a greater level of human activity due to their proximity to humans and the fact that they live in relatively small areas. There has been no attempt as yet to examine population structure for bottlenose dolphins in the Mediterranean Sea, beyond genetic evidence for a general split between the eastern and western Mediterranean, and with the Black Sea (Natoli *et al.* 2004). Figure 1.1.1 depicts the different assessment units currently proposed for European Atlantic populations. With more information, these will likely be modified. Population estimates for the small coastal populations have been derived mainly using capture-mark-recapture analytical techniques, where animals are “captured” photographically, and identified individually by unique markings mainly on the dorsal fin, such as patterns of nicks. The same or different individuals may then be “re-captured” photographically in successive encounters.

In northern Europe, the continental shelf edge is generally some distance from the coast thus separating coastal and offshore ecotypes spatially, although groups of the latter may on occasions enter coastal waters. Further south in the Bay of Biscay and around the Iberian Peninsula, the shelf edge comes close to the coast and it becomes more difficult to separate the two ecotypes unless individuals are recognized photographically. Population estimates in this region are mainly derived from line-transect surveys to determine absolute abundance rather than by photo-ID, and no attempt therefore has been made to try to distinguish between ecotypes or management units. Those population estimates are highlighted in red or orange in Figure 1.1.1.

In coastal waters, bottlenose dolphins often favour river estuaries, semi-enclosed bays, headlands and sandbanks where there is uneven bottom relief and/or strong tidal currents (Lewis & Evans 1993, Liret *et al.* 1994, Wilson *et al.* 1997, Liret 2001, Ingram & Rogan 2002).

Genetic studies using mitochondrial DNA indicated some population sub-structuring between animals from east and west Scotland and from Wales, although for the latter two regions, sample sizes were low; there was also a low level of genetic diversity in the east Scottish sample (Parsons *et al.* 2002). Using both mtDNA and microsatellite techniques, fine-scale population structure was revealed among three distinct populations in Ireland – one in the Shannon Estuary, another from the Connemara-Mayo region and a third from strandings of unknown origin but thought to be part of a large offshore population (Mirimin *et al.* 2011). This study found moderate (microsatellite) to low (mitochondrial) gene diversity in dolphins using the Shannon Estuary and the Connemara–Mayo region, while dolphins that stranded along the coast showed much higher levels of gene diversity at both classes of markers (Mirimin *et al.* 2011).

Further south, a genetic study by Fernández *et al.* (2009b) of stranded bottlenose dolphins in Galicia, NW Spain, using mtDNA and microsatellites, found significant differences between animals in northern and southern Galicia, as well as from Portugal. There were also significant differences in microsatellite frequencies between southern Galicia and the northeastern corner of Spain. However, most of these sample sizes were very small. These studies, as well as one by Nichols *et al.* (2009) using ancient DNA that found significant differentiation in a now extinct population from the Humber Estuary, Eastern England, suggest that local adaptation leading to isolation and potential extinction of coastal populations of bottlenose dolphin in Europe may be a feature of this species, exposing them to long-term conservation risk. Offshore populations, on the other hand, seem to exhibit much higher levels of gene flow (e.g. Madeira and the Azores – see Querouil *et al.* 2007).

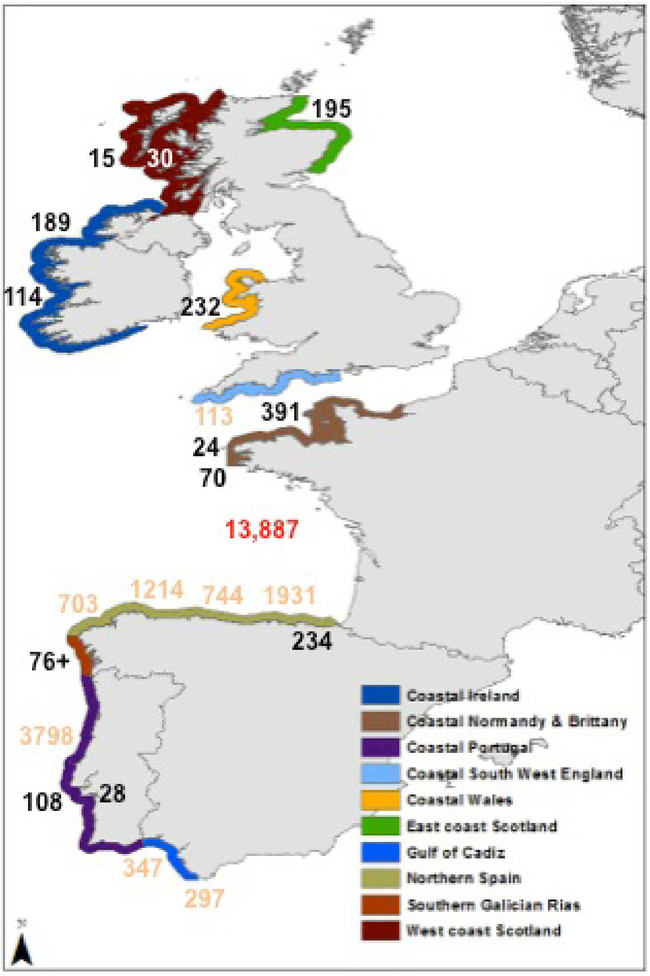


Figure 1.1.1 *Assessment Units and their population estimates for bottlenose dolphin in coastal Atlantic Europe. Values in red refer to populations believed to be largely of the offshore ecotype. Those in black are believed to be of the coastal ecotype, and the value in orange may be a mixture of the two*

Conclusions

With our current knowledge, the offshore bottlenose dolphin ecotype in the Atlantic could be listed under category S4 a population with individuals with inherently large home ranges forming one mixing population at the supra-national level, or in category MR1 as it has a clearly sustainable metapopulation above Member State level and is not generally considered to be a fully migratory species.

The coastal ecotype forms a number of management units (at least ten) in Atlantic Europe (MATL biogeographic region of ES, FR, IE, PT and UK) and likely forms others in the Mediterranean and Black Sea. Some of these occur across Member State boundaries (e.g. Channel Islands and French Normandy coasts) but most occur below Member State level, and sometimes are very localised (e.g. Shannon Estuary, Ireland; and Sado Estuary, Portugal). These would therefore most closely fit either category S1, populations (assessment units) with more or less exchange at or below national level, or possibly in a few cases (e.g. Black Sea), category S2, populations of large, more or less mobile sedentary species with only one or a few clearly isolated population(s) within a Member State.

Step 1.3 - Historical perspective: what happened to the species?

The common bottlenose dolphin was more widely distributed in the European mainland coastal zone in historical times (see Figure 1.1.2). Since the late 19th century, a number of coastal bottlenose dolphin populations, particularly those occupying estuaries, have declined or disappeared altogether. Bottlenose dolphins appear to have used some coastal areas for only limited periods of time, possibly forming ephemeral populations. For example, a group of dolphins utilised the Noirmourtier area (France) in the 1950-60s and similar reports have been made for the Quiberon-Houat-Hoedic area (France). It is unclear whether these were truly resident coastal populations or offshore visitors that remained in those areas for a limited period of time. The best recent example is use of the Marsdiep area and the area east of Texel (The Netherlands). The species was recorded there regularly, and in relatively large numbers (up to 30-40 at a time), between 1933 and 1939 by Verwey (1975), mainly between February and May, coinciding with the migration and spawning period of the Zuiderzee herring. After the closure of the Zuiderzee bay in 1932, the Zuiderzee herring gradually disappeared from the area, and in the late 1930s the regular occurrence of relatively large numbers of bottlenose dolphins ceased. Observations outside the Marsdiep area between the 1930s and 1970 are anecdotal, but the species was regarded as common in all Dutch waters and estuaries, second only to the harbour porpoise (Camphuysen & Peet 2006, Camphuysen & Smeenk 2016). After 1970, the species became scarce in Dutch waters, with strandings also declining rapidly (Kompanje 2001, Camphuysen & Peet 2006, Camphuysen & Smeenk 2016), at a similar time as a reduction in stranding records from SE England, as well as further west in SW England (Evans 1980, Tregenza 1992).

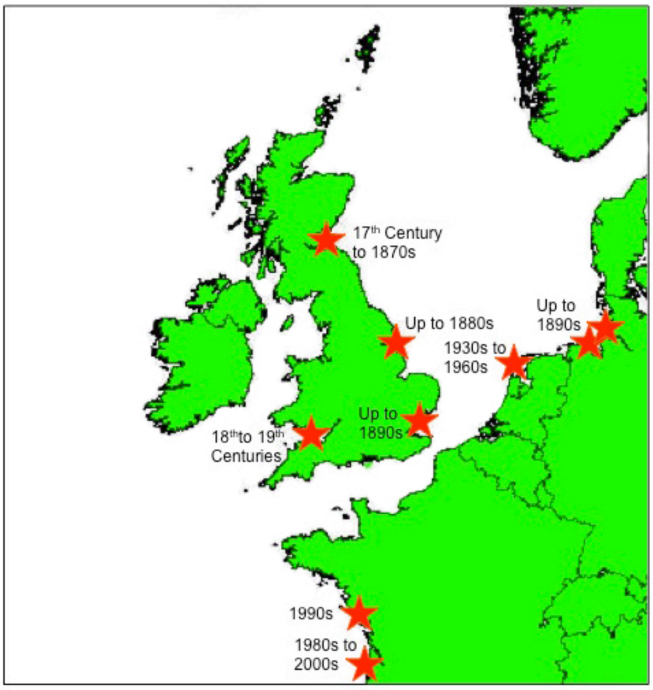


Figure 1.1.2 *Historical distribution of coastal bottlenose dolphins in NW Europe (from Evans & Scanlan 1990, ICES 2016)*

Earlier status changes are difficult to ascertain but historical accounts indicate that bottlenose dolphins occurred in the Severn Estuary, Thames Estuary, Humber Estuary, and Firth of Forth until the late nineteenth century (Evans & Scanlan 1990, Nichols *et al.* 2007). Along the coast of Germany, bottlenose dolphins occurred in the Elbe (Goethe 1983) and Weser estuaries (Mohr 1935, Goethe 1983, Kölmel & Wurche 1998) until the late nineteenth century. Further south, in Portuguese waters, bottlenose dolphins were reported in the Tagus estuary until 1960 (Teixeira 1979). More recently, a coastal group persisted at Arcachon (France) between the late 1980s until it disappeared in the early 2000s, and a group of six animals occurred at Pertuis Charentais, between Ré and Oléron Islands and the French mainland, for a period in the late 1990's.

In conclusion, currently, coastal bottlenose dolphins occur within each of the assessment units. In past centuries, the species occupied the southern North Sea and a number of estuaries where they now do not occur, or occur only as an uncommon visitor.

A number of human activities may affect bottlenose dolphins. The most obvious human pressure in coastal areas is human disturbance mainly from recreational activities (including commercial dolphin watching), with both short- and long-term impacts noted in several populations around the world (Bejder & Samuels 2003, Bejder *et al.* 2006) including West Wales (Feingold & Evans 2014a; Norrman *et al.* 2015) and East Scotland (Pirotta *et al.* 2014, 2015). Incidental bycatches of bottlenose dolphins through entanglement in fishing gear (mainly gillnets and pelagic trawls) also occur (ICES 2015a; b). Within the OSPAR region, bottlenose dolphin incidental bycatch appears to be highest (and potentially unsustainable) off the coasts of northern Spain (Galicia, Asturias, Cantabria and Basque Country), west Portugal, and SW Spain (Andalucia) (López *et al.* 2003, 2012, Goetz *et al.* 2014, Vázquez *et al.* 2014, Vélez 2014, ICES, 2015a). Fishing activities may also indirectly affect populations through depletion of the prey resource (ICES, 2015c). Habitat disturbance as a result of fishing activities causing damage to the seabed and its benthic faunas has been suggested as a human pressure in some areas, although this remains to be supported by evidence (Feingold & Evans, 2014a; Norrman *et al.*, 2015). Habitat loss has also affected coastal populations (Camphuysen & Peet 2006, Camphuysen & Smeenk 2016). Research has demonstrated high pollutant loads in most of the investigated coastal bottlenose dolphin populations, possibly leading to health issues and reproductive failure (Jepson *et al.* 2013, 2016). Exposure to high pollutant levels has been suggested as one reason for past declines and disappearance of some populations (Jepson *et al.* 2013, 2016). Climate change may also affect bottlenose dolphins, positively or negatively, either by altering human activities and thus pressures, or by affecting the stock sizes and distributions of their prey (Evans & Bjørge 2013), and increased underwater noise may have negative effects (Bailey *et al.* 2010).

Step 1.4 - Analysis of distribution and trends

The only large-scale abundance surveys in Europe have been those of SCANS (July 1994) and SCANS II (July 2005) in NW European shelf seas, and CODA (July 2007) surveying offshore largely beyond the NW European shelf break. SCANS II surveys (2005) gave an overall abundance estimate of 12,600 (CV=0.27; 95% CI: 7,500-21,300) bottlenose dolphins, with highest densities near the shelf break (Hammond *et al.* 2013), whereas the CODA survey (2007) beyond the shelf edge yielded an abundance estimate (uncorrected for g(0) and responsive movement), of 19,300 (CV=0.25; 95% CI: 11,900-31,300) (CODA 2009). These highlight the significant offshore populations of this species. The coastal Atlantic Europe populations are thought to number somewhere between 2-4,000 individuals (ICES, 2016; Figure 1.1.1). In July 2016, a further large-scale abundance survey, SCANS III was undertaken. Preliminary results have been published (ICES 2017) but are awaiting supplementary information to be made available from surveys in Irish waters (OBSERVE Programme) before complete results can be presented.

Of the available datasets, only five sites, all in the Atlantic region, have had sufficient repeated abundance estimates to allow any assessment of trends in abundance. For each assessment unit (AU), the estimated population size and trend, where known, are summarised below.

West coast of Scotland AU

A small resident bottlenose dolphin population numbering around 15 animals inhabits the vicinity of the Sound of Barra in the Outer Hebrides (Grellier & Wilson 2003, Cheney *et al.* 2013, Van Geel 2016) whilst an estimated 30 bottlenose dolphins range around the Inner Hebrides spending periods of time around Islay, the Small Isles, Skye, and occasionally the Minch north of Skye (Cheney *et al.* 2013, Van Geel 2016). There are insufficient data to determine the population trends at this time, although the numbers appear to be stable over the two last decades.

East coast of Scotland AU

Monitoring of bottlenose dolphins in the inner Moray Firth started in 1990, and later was extended to a wider part of the Firth. Even though bottlenose dolphins ranged all along the north and south coasts of the Moray Firth during the 1990s, it was not until the mid-1990s that the species started extending its range around the Grampian coast (Evans *et al.* 2003, Wilson *et al.*, 2004). It is now regularly encountered particularly off Aberdeen, the coast of Fife and in St Andrews Bay (Weir & Stockin 2001, Cheney *et al.* 2013). Bottlenose dolphins, some of which have been photo-identified as belonging to the Moray Firth population, are now seen annually along the coast of NE England as far south as Yorkshire (Sea Watch Foundation, unpublished data).

Analysis of mark-recapture studies using a Bayesian approach estimates the population on the East Coast of Scotland at 87-208 animals, with the latest estimate (2014) being 170 (95% Highest Posterior Density interval: 139-200). Despite inter-annual variability, the population is considered to be stable or increasing, with no decline over the available time series of >5% in 10 years (Figure 1.1.3; Cheney *et al.* 2013).

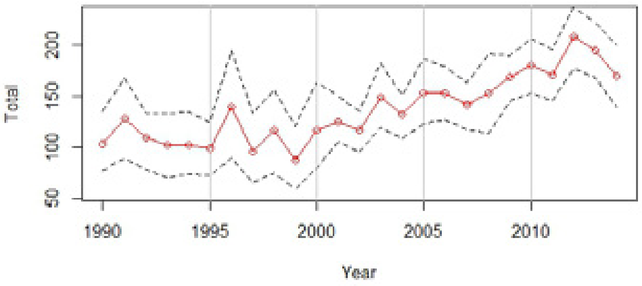


Figure 1.1.3 *Population trend, East Coast of Scotland AU*

Although bottlenose dolphins are occasionally recorded offshore in the North Sea and in coastal waters off SE England, northern France, Belgium, The Netherlands and Germany, there is no evidence that these are anything but transient animals, most likely from the East Coast of Scotland population or from further afield (Evans *et al.* 2003, Camphuysen & Peet 2006, Evans & Teilmann 2009, ICES 2013).

Coast of Wales AU

Annual monitoring of bottlenose dolphins in Cardigan Bay Special Area of Conservation, West Wales (UK), began in 2001. This was extended to incorporate the wider Cardigan Bay area from 2005. In addition, since 2007, there have been opportunistic photo-ID surveys in the coastal waters of North Wales, and occasionally around the Isle of Man and in Liverpool Bay (Pesante *et al.* 2008, Feingold & Evans 2014a, Norrman *et al.* 2015). A proportion of the population inhabiting Cardigan Bay in summer ranges more widely between November and April, occurring particularly off the north coast of Anglesey, the mainland coast of North Wales and further north around the Isle of Man (Feingold & Evans 2014b). Summer mark-recapture estimates for Cardigan Bay SAC have varied from 116-260 animals. The latest estimate (2015) is 159 (95% CI = 130-228) animals. For the wider Cardigan Bay (including both SACs), summer mark-recapture estimates have varied from 152-342 animals, with the 2015 estimate being 222 (95% CI: 184-300) animals. The Coastal Wales Assessment Unit population is considered to be stable, with no decline over the available time series of >5% in 10 years (Figures 1.1.4 & 1.1.5). It is noted that between 2013 and 2015, the population estimates have been amongst the lowest recorded but due to variability in the estimates it is too early to determine whether this indicates a decline.

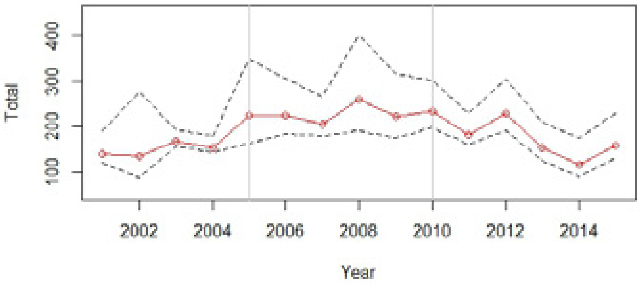


Figure 1.1.4 *Population trend, Cardigan Bay SAC (Coast of Wales AU)*

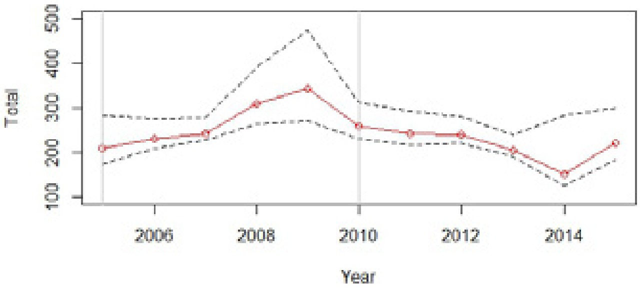


Figure 1.1.5 *Population trend, Wider Cardigan Bay (Coast of Wales AU)*

Coast of Ireland AU

Bottlenose dolphins are regularly recorded in several bays along the west coast of Ireland, notably Kenmare River and Brandon Bay (Co. Kerry), Clew Bay and adjacent coastal areas of Connemara (Co. Galway), Broadhaven Bay (Co. Mayo), and Donegal Bay (Co. Donegal) (Ingram *et al.* 2001, 2003, Ó Cadhla *et al.* 2003, Evans *et al.* 2003). They have also been recorded all along the south coast of Ireland, with sightings mainly around Cork Harbour (Co. Cork) and Rosslare Harbour (Co. Wexford) (Evans *et al.* 2003; O'Brien *et al.* 2009). Photo-ID matches indicate that individual bottlenose dolphins may range all around the coast of Ireland, and although there is a more or less continuous distribution from inshore to offshore, there is both photo-ID and genetic evidence for an offshore ecotype west of Ireland (O'Brien *et al*, 2009, Mirimin *et al.* 2011, Oudejans *et al.* 2015). There are a number of mark-recapture population estimates for animals using the west coast of Ireland, but at different spatial scales. One estimate for NW Connemara is 171 individuals (95% CI: 100-294) in 2009 (Ingram *et al.* 2009) and a second estimate for a much larger area, including Connemara, Mayo and Donegal, numbered 151 (95% CI: 140–190) individuals for the year 2014 (Nykanen *at al*. 2015). This mobile population appears to range widely, with seasonal and patchy habitat use. There is not enough information to indicate population trends.

Bottlenose dolphins inhabit the Shannon Estuary year-round, and genetic studies indicate that they form a discrete population separate from those occurring elsewhere along the west coast of Ireland (Mirimin *et al.* 2011). Six mark-recapture population estimates produced between 1997 and 2015 range from 107 to 140 individuals (Ingram 2000, Ingram *et al.* 2008, Berrow *et al.* 2010). The latest population estimate (2015) is 114 (95% CI: 90-143) (Rogan *et al.* 2015) indicating that the population is probably stable (Figure 1.1.6).

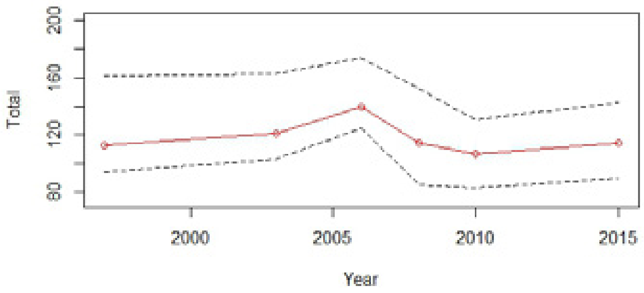


Figure 1.1.6 *Population trend, Shannon Estuary (Coast of Ireland AU)*

Coast of Southwest England AU
Bottlenose dolphins have regularly inhabited the south and southwest coasts of England since the 1990s, being most common around Cornwall but rare east of Dorset (Williams *et al.* 1998, Evans *et al.* 2003, Brereton *et al.* in review). No systematic photo-ID surveys have been undertaken, but Brereton *et al.* (in review) have reported maximum abundance estimates for south-west England coastal waters, using two mark-recapture methods, ranging between 102 and 113 (95% CI: 87-142) individuals over the combined period 2008-2013. There are insufficient data to assess trends.

Coast of Normandy and Brittany AU (France, UK)
A resident population of bottlenose dolphins inhabits the Gulf of St Malo, ranging between the French coast of Normandy and the Channel Islands (Couet 2015a, b, Louis *et al.* 2015). Mark-recapture estimates of this population in 2010 indicated it numbering between 372 (95% CI = 347-405) and 319 (95% CI= 310-327) animals, with a 2014 estimate of 340 (95% CI = 290-380) animals (Couet 2015a, b, Louis *et al.* 2015), thus indicating no significant difference (Figure 1.1.7).

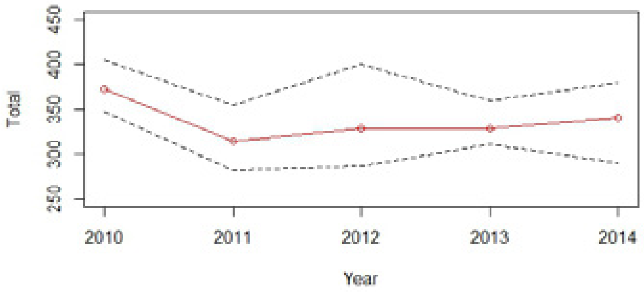


Figure 1.1.7 *Population trend, Gulf of St Malo (Coast of Normandy and Brittany AU)*

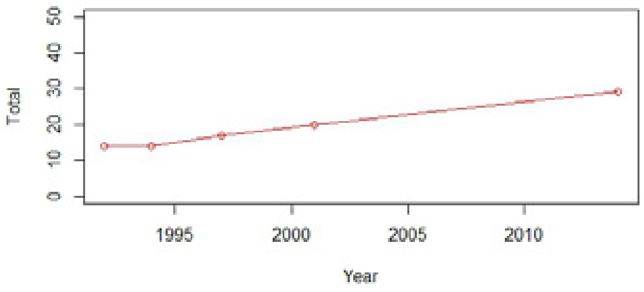


Figure 1.1.8 *Population trend for Ile de Seine (Coast of Normandy and Brittany AU)*

Two small populations, which appear to be distinct, exist in the Iroise Sea, one around Ile de Sein and the other around the Molène Archipelago. Photo-ID surveys have been undertaken in the vicinity of Ile de Sein since 2001, with at least five separate counts, ranging from 14 individuals in 2001 to 29 in 2014. An earlier estimate for this population was 14 animals in 1992 (Liret 2001, Liret *et al.* 2006), thus indicating a steady increase (Figure 1.1.8).

Around the Molène Archipelago, a mark-recapture estimate of 29 individuals (95% CI = 28-42) was produced from photographs taken between 1999 and 2001 (Le Berre & Liret 2001, Liret *et al.* 2006, Louis & Ridoux 2015). A new photo-ID analysis is currently being undertaken (V. Ridoux *pers. comm*.). It is therefore currently not possible to assess trends in this population.

Northern Spain

In northern Spanish waters, only model-based abundance estimates exist, derived from line-transect surveys conducted between 2003 and 2011. These encompass both coastal and offshore animals (López *et al.* 2013). The annual uncorrected abundance estimate in the study area is 10,687 individuals (95% CI of 4,094-18,132). Estimated abundances for the different areas are: (1) Euskadi: 1,931, (2) Cantabria: 744, (3) Asturias: 1,214, (4) Galicia: 703, (5) Galician Bank: 108 and (6) Aviles: 234. Although the distribution is homogeneous throughout the northern peninsula, there is a clear gradient in density, this being higher in eastern areas of the Bay of Biscay where the largest groups have been recorded (López *et al.* 2013). There are insufficient data at this time to make an assessment of trends.

Southern Galician Rias (North-west Spain)

Along the Galician coast, photo-ID surveys have been conducted between 2006 and 2009, resulting in the identification of 255 individuals (García *et al.* 2011). A third of these photo-identified individuals (n=76) were considered to form the resident population inhabiting the Southern Galician Rias, as revealed by recapture histories, genetics and stable isotope analysis (Fernández *et al.* 2011a, b, García *et al.* 2011). Movements of individuals were recorded between Galicia and Euskadi in the Bay of Biscay (García *et al.* 2011). It is not possible to make an assessment of trends in this population at this time.

Coast of Portugal

Bottlenose dolphins occur widely along the coast of Portugal as well as further offshore. Photo-ID surveys undertaken over two time periods have been used to derive mark-recapture population estimates of bottlenose dolphins in coastal Setúbal Bay (Martinho 2012, Martinho *et al.* 2015). Bottlenose dolphins identified from 1998-2001 were considered a closed and a more cohesive group than those from 2007-2011, with stable associations and an abundance of 106 (95% CI: 69–192) individuals. The more recent animals sampled seemed to be composed of an open group of 108 (95% CI: 83–177) animals, with a migration rate of 19% per year and low association values.

A wider-scale analysis of animals photographed in central west coastal Portugal from Nazaré and Sétubal Bay between 2008 and 2014 resulted in an estimate of 352 individuals (95% CI: 294-437) (Martinho 2012, Martinho *et al.* 2015).

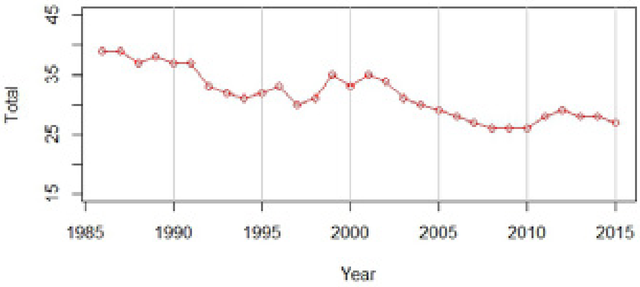


Figure 1.1.9 *Population trend, Sado Estuary (Coast of Portugal)*

There have been a number of line-transect surveys by both ship and plane undertaken off West Portugal between 2010 and 2014, covering the region between the coast and 50 nautical miles offshore. The estimated abundance for 2010-14 from aerial surveys was 2,306 animals (34.7% CV), and in 2011, using vessel surveys, it was 3,798 animals (87.6% CV) (Araújo *et al.* 2014). It is not possible to distinguish between the coastal population of this Assessment Unit and the offshore population.

The longest sequence of counts for a coastal bottlenose dolphin population in Europe is associated with the resident population in the Sado Estuary, where an annual census has been undertaken since 1986 (Gaspar 2003, Lacey 2015). Over this period, the population has shown a long-term decline from 39 individuals in 1986 to 28 individuals in 2014 (Figure 1.1.9), with pollution of the estuary proposed as a possible background for the decline (Van Bressem *et al.* 2003).

Gulf of Cadiz (Spain)
Mark-recapture estimates for bottlenose dolphins in the coastal Gulf of Cadiz have been determined for two periods: 2005-2006 and 2009-2010 (MAGRAMA 2012). These gave estimates of 347 individuals (95% CI: 264-503) for 2005-2006, and 397 (95% CI: 300-562) for 2009-2010 suggesting no significant difference. A much larger population apparently occupies the offshore Gulf of Cadiz, estimated to number 4,391 animals (95% CI: 2,373-8,356) in 2009-2010 (MAGRAMA 2012). It is not possible to make an assessment of trends at this time.

A bottlenose dolphin population also inhabits the area around the Strait of Gibraltar, on the edge of Area IV. Photo-ID surveys in 2010 resulted in a mark-recapture population estimate of 297 individuals (95% CI: 276-332) (Portillo *et al.* 2011). It is not possible to make an assessment of trends at this time.

Conclusions

In several areas within Europe (as has been found elsewhere in the world), coastal bottlenose dolphins are thought to form a number of demographically independent populations commonly referred to as management (or assessment) units. These have only been established within the Atlantic Region. Population trends for these should therefore be considered separately. Within the AUs, most of the available data relating to abundance (and therefore for examining trends) are from photo-identification studies of small localised resident or semi-resident groups, which are often related to monitoring the numbers of animals in protected areas. Evidence of trends was evaluated by examining abundance estimates at specific sites within the AUs where sufficient data exist (i.e., at least four abundance estimates from different years over a ten-year time period). No estimates exist for any site before 1986, and for most, estimates started only after the Habitats Directive came into force. Thus, there are no data prior to human impacts in these areas, so it is not possible to set a historical baseline. Although the historical abundance and distribution is unknown, there is good evidence that the species was once more widely distributed around these coasts. The recent trends available for six sites indicate little change over a ten-year period, with the exception of the small isolated population in the Sado Estuary, Portugal, which has shown a steady decline since the mid-1980s.

Step 2.1 - FRP assessment
Virtually no quantitative information exists of population numbers for bottlenose dolphin before the Habitats Directive came into force in 1992, and it should be noted that this species is actually better known than most other cetaceans. Nevertheless, there is strong evidence that the species was more widely distributed (and therefore probably more abundant) in the European mainland coastal zone, with regular occupation of a number of estuaries in historical times (see step 1.3). The indication therefore is that FRP for several coastal assessment units should be above the current values. The common argument used for setting FRPs at larger values to those obtained from recent abundance estimates is that environmental conditions have changed since so that the carrying capacity for a local population may also be lower than any FRP assessment. On the other hand, there is evidence that if conditions are improved, areas can be re-colonised (as, for example, appears to be occurring around the Firth of Forth, East Scotland). With some understanding of habitat preferences and food resource needs, however, one can compare against relatively undisturbed extant populations, and use that information as a baseline. This would best be undertaken by assessment unit where possible.

For the offshore ecotype, one could use the best estimate of Current Value as the FRP. For the coastal ecotype and the various assessment units within it, historical baselines would be preferable. These would best be derived by genetic analysis to estimate Ne, genetic diversity, population history (evidence of bottlenecking), and degree of connectivity between AUs. The order of magnitude of FRP cannot be determined until the genetic analyses described above have been undertaken. It is likely that the Ne/N ratio for coastal bottlenose dolphin is less than has been applied for mainly terrestrial species, but in any case, levels of connectivity between populations need to be better assessed first.

Conclusions

For the offshore ecotype within MATL FRP=CV

For the coastal ecotypes within MATL FRPs for the assessment units (AUs) are unknown but can be derived by genetic analysis to estimate Ne (effective population size), genetic diversity, population history (evidence of bottlenecking), and degree of connectivity between AUs. 'Guestimates' what the FRV might be for each AU in terms of broad ranges of population estimates are:

West coast of Scotland AU: 50-500
East coast of Scotland AU: 150-500
Coast of Wales AU: 250-500
Coast of Ireland AU: 300-1000 (Shannon Estuary: 100-200)
Coast of Southwest England AU: 100 -300
Coast of Normandy & Brittany AU: 500-1000
Sado Estuary AU: 30-50

Step 2.2 - FRR assessment

As described earlier, coastal bottlenose dolphin populations once existed in most of the major estuaries and bays of Western Europe. The degree to which they were permanent residents is difficult to ascertain historically, although in the last century, some areas (e.g. southern North Sea) had populations regularly living there for periods lasting at least some decades. Given that the species is highly mobile, there is potential for the locations currently without bottlenose dolphin populations, to be re-colonised, so long as there are suitable environmental conditions to sustain adequate prey resources. As a good example, the population normally inhabiting the Moray Firth in Northeast Scotland has been extending its range along the coast since the 1990s (ICES, 2016). In the last ten years, bottlenose dolphins identified as from the Moray Firth population have been seen annually in Eastern England, and in summer 2016, the species was even recorded off the coast of Norfolk in the southern North Sea. Up to now, these have been transient movements, but if conditions were suitable, there is potential for the species to re-occupy the southern North Sea as a resident.

Conclusions

For the offshore ecotype within MATL FRR=CV

For the coastal ecotypes within MATL FRRs for the assessment units (AUs) are unknown but can be assessed when FRPs are available.

References

Ansmann, I.C., Parra, G.J., Lanyon, J.M., and Seddon, J.M. (2012) Fine-scale genetic population structure in a mobile marine mammal: inshore bottlenose dolphins in Moreton Bay, Australia. *Molecular Ecology*, 21: 4472–4485.

Araújo, H., Santos, J., Rodrigues, P., Vingada, J., Eira, C., Raínho, A., Arriegas, I., Leonardo, T., Nunes, M., and Sequeira, M. (2014). *First technical proposal of new SCI sites dedicated to the conservation of cetaceans in Portugal for inclusion in the National List of Sites*. Annex to the 2014 Report of project LIFE MArPro NAT/PT/00038.

Bailey, H., Senior, B., Simmons, D., Rusin, J., Picken, G., and Thompson, P.M. (2010) Assessing underwater noise levels during pile-driving at an offshore windfarm and its potential effects on marine mammals. Marine Pollution Bulletin 60: 888–897.

Bejder, L. and Samuels, A. (2003) Evaluating the effects of nature based tourism on cetaceans. In: Gales, N., Hindell, M., and Kirkwood, R. (eds.) *Marine Mammals and Humans: Towards a Sustainable Balance*. Collingwood, Australia: CSIRO Publishing, pp. 229-256.

Bejder, L., Samuels, A., Whitehead, H., Gales, N., Mann, J., Connor, R.C., Heithaus, M.R., Watson-Capps, J., Flaherty, C., and Krutzen, M. (2006) Decline in relative abundance of bottlenose dolphins exposed to long-term disturbance. *Conservation Biology*, 20: 1791-1798.

Berrow, S., O'Brien, S., Groth, L., Foley, A., and Voigt, K. (2010) *Bottlenose Dolphin SAC Survey.* Report to the National Parks and Wildlife Service**.** 24pp.

Bilgmann, K, Möller, L.M., Harcourt, R.G., Gibbs, S.E., and Beheregaray, L.B. (2007). Genetic differentiation in bottlenose dolphins from South Australia: association with local oceanography and coastal geography. *Marine Ecology Progress Series*, 341: 265–276.

Brereton, T., Jones, D., Leeves, K., Lewis, K., Davies, R., and Russell, T. (in review) Population structure, mobility and conservation of Common Bottlenose Dolphin off southwest England from photo-identification studies. *Journal of the Marine Biological Association of the United.Kingdom.*

Camphuysen C.J. and G. Peet (2006) *Whales and dolphins in the North Sea*. Fontaine Uitgevers, Kortenhoef, The Netherlands.

Camphuysen, C.J. and Smeenk, C. (2016) Tuimelaar *Tursiops truncatus*. In: S. Broekhuizen, J.B.M. Thissen, K.J. Canters & J.C. Buys (eds). *Atlas van de Nederlandse zoogdieren. – Natuur van Nederland 12*. Naturalis Biodiversity Center / European Invertebrate Survey, Leiden.

Certain, G., Ridoux, V., Van Canneyt, O. and Bretagnoile, V. (2008) Delphinid spatial distribution and abundance estimates over the shelf of the Bay of Biscay. ICES Journal of Marine Science, 65, 656-666.

Chilvers, B.L., and Corkeron, P.J. (2001) Trawling and bottlenose dolphins' social structure. *Proceedings of the Royal Society of London*, Series B 268: 1901–1905.

Cheney, B.J., Thompson, P.M., Ingram, S.N., Hammond, P.S., Stevick, P.T., Durban, J.W., Culloch, R.M., Elwen, S.H., Mandleberg, L., Janik, V.M., Quick, N.J., Islas-Villanueva, V., Robinson, K.P., Costa, M., Eisfeld, S.M., Walters, A., Philips, C., Weir, C.R., Evans, P.G.H., Anderwald, P., Reid, R.J., Reid, J.B., and Wilson, B. (2013) Integrating multiple data sources to assess the distribution and abundance of bottlenose dolphins (*Tursiops truncatus*) in Scottish waters. *Mammal Review*, 43: 71-88.

Cheney, B., Corkrey, R., Durban, J.W., Grellier, K., Hammond, P.S., Islas-Villanueva, V., Janik, V.M., Lusseau, S.M., Parsons, K.M., Quick, N.J., Wilson, B. and Thompson, P.M. (2014) Long-term trends in the use of a protected area by small cetaceans in relation to changes in population status. *Global Ecology and Conservation,* 2: 118-128.

CODA (2009) Cetacean Offshore Distribution and Abundance in the European Atlantic (CODA). Sea Mammal Research Unit, St Andrews. 43pp.

Couet, P. (2015a) From identification to models: reassessment of usual methods for populations monitoring. The case of bottlenose dolphin (*Tursiops truncatus*) inhabiting the English Channel. Poster presented at the international Workshop « Bottlenose dolphin conservation and monitoring in the North-western Mediterranean Sea » in Marseille, France.

Couet, P. (2015b) *De l'identification des animaux aux modèles mathématiques: une remise en question des méthodes usuelles de suivi des populations. Le cas de la population de grands dauphins (*Tursiops truncatus*) en mer de la Manche*. MSc thesis, Centre d'Ecologie Functionnelle & Evolutive.

Curry, B.E., and Smith, J. (1998) Phylogeographic structure of the bottlenose dolphin (*Tursiops truncatus*): stock identification and implications for management. In: Molecular Genetics of Marine Mammals (editors Dizon, A.E., Chivers, S.J., and Perrin, W.F.), pp. 227–247. The Society for Marine Mammalogy, Special Publication 3, Lawrence, Kansas.

Englund, A., Ingram, S. and Rogan, E. (2007) Population status report for bottlenose dolphins using the Lower River Shannon SAC, 2006–2007. Final report to the National Parks and Wildlife Service.

Evans, P.G.H. (1980) Cetaceans in British Waters. *Mammal Review*, 10: 1-52.

Evans, P.G.H. and Bjørge, A. (2013) Impacts of climate change on marine mammals. Marine Climate Change Impacts Partnership (MCCIP) Science Review 2013, 134-148. Published online 28 November 2013 doi:10.14465/2013.arc15.134-148.

Evans, P.G.H. and Scanlan, G. (1990) Historical Status Changes of Cetaceans in British and Irish waters. *European Research on Cetaceans*, 3: 51-57.

Evans, P.G.H. and Teilmann, J. (Editors) (2009) Report of ASCOBANS/HELCOM Small Cetacean Population Structure Workshop. ASCOBANS/UNEP Secretariat, Bonn, Germany. 140pp.

Evans, P.G.H., Anderwald, P., and Baines, M.E. (2003) *UK Cetacean Status Review* Report to English Nature and the Countryside Council for Wales. Sea Watch Foundation, Oxford. 160pp.

Feingold, D. and Evans, P.G.H. (2014a) *Bottlenose Dolphin and Harbour Porpoise Monitoring in Cardigan Bay and Pen Llyn a'r Sarnau Special Areas of Conservation 2011-2013*. Natural Resources Wales Evidence Report Series No. 4. 124 pp.

Feingold, D. and Evans, P.G.H. (2014b) *Connectivity of Bottlenose Dolphins in Welsh Waters: North Wales Photo-Monitoring Report.* Natural Resources Wales Research Report. 15pp.

Fernández, R., Garcia-Tiscar, S., Santos, M.B., López, A., Martinez-Cedeira, J.A., Newton, J., and Pierce, G.J. (2011b). Stable isotope analysis in two sympatric populations of bottlenose dolphins *Tursiops truncatus*: evidence of resource partitioning? *Marine Biology*, 158: 1043-1055.

Fernández, R., Santos, M.B., Pierce, G.J., Llavona, A., López, A., Silva, M.A., Ferreira, M., Carrillo, M., Cermeño, P., Lens, S., and Piertney, S.B. (2011a) Fine-scale genetic structure of bottlenose dolphins, *Tursiops truncatus*, in Atlantic coastal waters f the Iberian Peninsula. *Hydrobiologica,* 670: 111-125.

Fernández, R., Garcia-Tiscar, S., Santos, M.B., López, A., Martinez-Cedeira, J.A., Newton, J., and Pierce, G.J. (2011b) Stable isotope analysis in two sympatric populations of bottlenose dolphins *Tursiops truncatus*: evidence of resource partitioning? *Marine Biology*, 158: 1043-1055.

García, N., Caldas, M., Palacios, G., Moldes, M., Leal, A., and López, A. (2011) Photo-identification and population movements of bottlenose dolphins *Tursiops truncatus*, along the Galician coast, NW Iberia. Poster, 25th Annual Conference of the European Cetacean Society, 21-23 March 2011, Cadiz, Spain.

Gaspar, R. (2003) *Status of the Resident Bottlenose Dolphin Population in the Sado Estuary: Past, Present and Future*. Doctoral Thesis. University of St. Andrews, Scotland. 194pp.

Goethe, F. (1983) Wale und Delphine in niedersächsischen Küstengewässern und Flüssen, Drosea, '83: 49-68.

Goetz, S., Read, F.L., Santos, M.B., Pita, C., and Pierce, G.J. (2014) Cetacean-fishery interactions in Galicia (NW Spain): results and management implications of a face-to-face interview survey of local fishers. *ICES Journal of Marine Science* 71(3): 604-617.

Grellier, K. and Wilson, B. (2003) Bottlenose dolphins using the Sound of Barra Scotland. *Aquatic Mammals*, 29: 378–382.

Hammond, P.S., Macleod, K., Berggren, P., Borchers, D.L., Burt, M.L., Cañadas, A., Desportes, G., Donovan, G.P., Gilles, A., Gillespie, D., Gordon, J., Hiby, L., Kuklik, I., Leaper, R., Lehnert, K., Leopold, M., Lovell, P., Øien, N., Paxton, C.G.M., Ridoux, V., Rogan, E., Samarra, F., Scheidat, M., Sequeira, M., Siebert, U., Skov, H., Swift, R., Tasker, M.L., Teilmann, J., Van Canneyt, O. and Vázquez, J.A. (2013) Cetacean abundance and distribution in European Atlantic shelf waters to inform conservation and management. *Biological Conservation,* 164**:** 107–122.

Hoban, S., Amtzen, J.A., Bruford, M.W., Godoy, J.A., Hoelzel, A.R. Segelbacker, G., Vila, C., and Bertorelle, G. (2014) Comparative evaluation of potential indicators and temporal sampling protocols for monitoring genetic erosion. *Evolutionary Applications*, 7(9): 984–998. http://dx.doi.org/10.1111/eva.12197.

Hoelzel, A.R. (1998) Genetic structure of cetacean populations in sympatry, parapatry and mixed assemblages; implications for conservation policy. *Journal of Heredity,* 89: 451–458.

Hoelzel, A.R., Potter, C.W., and Best, P.B. (1998) Genetic differentiation between parapatric 'nearshore' and 'offshore' populations of the bottlenose dolphin. *Proceedings of the Royal Society of London. Series B*, Biological Sciences, 265: 1177**–**1183.

ICES (2013) Report of the Working Group on Marine Mammal Ecology (WGMME), 4–7 February 2013, Paris, France. ICES CM 2013/ACOM:26. 117pp.

ICES (2015a) Bycatch of small cetaceans and other marine animals – Review of national reports under Council Regulation (EC) 812/2004 and other published documents. In: Report of the ICES Advisory Committee, 2015. ICES Advice 2015, Book 1, Section 1.6.1.1.

ICES (2015b) Report of the Working Group on Bycatch of Protected Species (WGBYC), 2-6 Feb 2015, ICES Headquarters, Copenhagen, Denmark. ICES CM 2015\ACOM:26. 82pp.

ICES (2015c) Report of the Working Group on Marine Mammal Ecology (WGMME), 9–12 February 2015, London, UK. ICES CM 2015/ACOM: 25. 114pp.

ICES (2016) 1.6.6.2 OSPAR request on indicator assessment of coastal bottlenose dolphins. ICES Special Request Advice. Northeast Atlantic Ecoregion. 14pp.

ICES (2017) Report of the Working Group on Marine Mammal Ecology (WGMME), 6–9 February 2017, St Andrews, UK. ICES CM 2017/ACOM: 27. 110pp

Ingram, S.N. (2000) *The ecology and conservation of bottlenose dolphins in the Shannon Estuary,* *Ireland.* PhD thesis, University College Cork, Ireland. 213pp.

Ingram, S.N. and Rogan, E. (2002) Identifying critical areas and habitat preferences of bottlenose dolphins *Tursiops truncatus*. *Marine Ecology Progress Series*, 244: 247-255.

Ingram, S. and Rogan, E. (2003). *Estimating abundance, site fidelity and ranging patterns of bottlenose dolphins (Tursiops truncatus) in the Shannon Estuary and selected areas of the west-coast of Ireland.* Report to the National Parks and Wildlife Service**.** 28pp.

Ingram, S.N., Englund, A., and Rogan, E. (2001). *An extensive survey of bottlenose dolphins (Tursiops truncatus) on the west coast of Ireland*. Heritage Council Report no. WLD/2001/42 17pp.

Ingram, S.N., Englund, A., and Rogan, E. (2003). *Habitat use, abundance and site-fidelity of bottlenose dolphins (Tursiops truncatus) in Connemara coastal waters, Co. Galway.* Heritage Council Wildlife Grant #12314. 27pp.

Ingram, S., Kavanagh, A., Englund, A., and Rogan, E. (2009) *Site assessment of the waters of northwest Connemara. A survey of bottlenose dolphins* (Tursiops truncatus). Report to the National Parks and Wildlife Service**.** 33pp.

Jepson, P.D., Deaville, R., Barnett, J., Davison, N.I., Paterson, I.A.P., Penrose, R.S., Perkins, M.P., Reid, R.J., and Law, R.J. (2013). Persistent organic pollutants in UK stranded cetaceans (1990-2008): linking exposure with toxic effects. In: P.G.H. Evans (ed.) *Chemical Pollution and Marine Mammals*. European Cetacean Society Special Publication 55: 18-24.

Jepson, P.D., Deaville, R., Barber, J.L., Aguilar, A., Borrell, A., Murphy, S., Barry, J., Brownlow, A., Barnett, J., Berrow, S., Cunningham, A.A., Davison, N.J., ten Doeschate, M., Esteban, R., Ferreira, M., Foote, A.D., Genov, T., Giménez, J., Loveridge, J., Llavona, A., Martin, V., Maxwell, D.L., Papachlimitzou, A., Penrose, R., Perkins, M.W., Smith, B., de Stephanis, R., Tregenza, N., Verborgh, P., Fernandez, A., and Law, R.J. (2016). PCB pollution continues to impact populations of orcas and other dolphins in European waters. *Nature Scientific Reports*. | 6:18573 | DOI: 10.1038/srep18573.

Kölmel, R. and Wurche, B. (1998) Wale aus der Elbe: Die Arten und ihre Fund- und Fangberichte. *Schriftenreihe Natureum Niederelbe*, 2: 10-35.

Kompanje, E.J.O. (2001) Review of strandings and catches of *Tursiops truncatus* (Mammalia: Cetacea, Odontoceti) in the Netherlands between 1754 and 2000. *Deinsea,* 1(8): 169-224.

Krützen, M., Sherwin, W.B., Berggren, P. and Gales, N. (2004) Population structure in an inshore cetacean revealed by microsatellite and mtDNA analysis: bottlenose dolphins (*Tursiops* sp.) in Shark Bay, Western Australia*. Marine Mammal Science*, 20: 28–47.

Lacey, C. (2015) *Current status of the resident bottlenose dolphin population in the Sado Estuary, Portugal*. Unpublished MSc dissertation, University of Edinburgh.

Lewis, E.J. and Evans, P.G.H. (1993) Comparative ecology of bottle-nosed dolphins (*Tursiops truncatus)* in Cardigan Bay and the Moray Firth. *European Research on Cetaceans,* 7: 57-62.

Liret, C. (2001) *Domaine vital, utilisation de l'espace et des ressources : les grands dauphins, Tursiops truncatus, de l'île de Sein*. PhD thesis, Université de Bretagne Occidentale, Brest. 155pp.

Liret, C., Allali, P., Creton, P., Guinet, C., and Ridoux, V. (1994) Foraging activity pattern of bottlenose dolphins around Ile de Sein, France, and its relationships with environmental parameters. *European Research on Cetaceans, 8*: 188-191.

Liret, C., Baines, M.E., Evans, P.G.H., Hammond, P.S. and Wilson, B. (2006) Atlantic bottlenose dolphins: conservation and management. Oceanopolis, Brest, France. 56pp.

Liret, C., Creton, P., Evans, P.G.H., Heimlich-Boran, J.R. and Ridoux, V. (1998) English and French coastal Tursiops from Cornwall to the Bay of Biscay, 1996. Photo-Identification Catalogue. Project sponsored by Ministère de L'Environnement, France and Sea Watch Foundation, UK. 100pp.

López, A., Pierce, G.J., Santos, M.B., Gracia, J., and Guerra, A. (2003) Fishery bycatches of marine mammals in Galician waters: results from on-board observations and an interview survey of fishermen. *Biological Conservation*, 111: 25-40.

López, A., Veacquez, J.A., Martínez-Cedeira, J.A., Marcos-Ipiña, E., Laria, L., Maestre, I., Carbo, A., Llanova, A., Fernández, M., Díaz, J.L., Santos, L., Ruano, A., Fernández, R., y Méndez, P. (2012) *Bases para el desarrollo de los planes de conservación de las espaceies de cétaceos protegidas en la Demarcación Marina Noratlantica. Volumen 2 Impactos*. Ministerio de Agricultura, Alimentacíon y Medio Ambiente-Fundación Biodiversidad.

López, A., Vázquez, J. A., Martínez, J., Cañadas, A., Marcos, E., Maestre, I., Ruano, A., Laria, L., Llanova, A., Macleod, K. and Evans, P. (2013) New abundance estimates for harbour porpoise (*Phocoena phocoena*) and bottlenose dolphin (*Tursiops truncatus*) in Northern Spanish Cantabrian and adjacent waters of Bay of Biscay (2003–2011). Poster presented at the 27th European Cetacean Society Annual Conference, 8th–10th April, 2013, Setúbal, Portugal.

Louis, M. and Ridoux, V. (2015) *Suivi des grands dauphins et des petits cétacés dans le Parc naturel marin d'Iroise*. Rapport Convention n° 920/UMS 3462/PNMI.

Louis, M., Fontaine, M.C., Spitz, J., Schlund, E., Dabin, W., Deaville, R., Caurant, F., Cherel, Y., Guinet, C. and Simon-Bouhet, B. (2014a) Ecological opportunities and specializations shaped genetic divergence in a highly mobile marine top predator. Proc. R. Soc. B 281: 20141558. http://dx.doi.org/10.1098/rspb.2014.1558

Louis, M., Gally, F., Barbraud, C., Béesau, J., Tixier, P., Simon-Bouhet, B., Le Rest, K., and Guinet, C. (2015) Social structure and abundance of coastal bottlenose dolphins, *Tursiops truncatus*, in the Normano-Breton Gulf, English Channel. *Journal of Mammalogy*, DOI:10.1093/jmamma/gyv053.

Louis, M., Viricel, A., Lucas, T., Peltier, H., Alfonsi, E., Berrow, S., Brownlow, A., Covelo, P., Dabin, W., Deaville, R., de Stephanis, R., Gally, F., Gauffier, P., Penrose, R., Silva, M. A., Guinet, C. and Simon-Bouhet, B. (2014b) Habitat-driven population structure of bottlenose dolphins, *Tursiops truncatus*, in the North-East Atlantic. *Molecular Ecology;*, 23: 857–874.

MAGRAMA (2012) *Estrategias Marinas. Grupo Mamíferos Marinos. Evaluación inicial y buen estado ambiental.* Technical report*.*

Martinho, F., Pereira, A., Brito, C., Gaspar, R., and Carvalho, I. (2015) Structure and abundance of bottlenose dolphins (*Tursiops truncatus*) in coastal Setúbal Bay, Portugal. *Marine Biology Research,* 11(2): 144-156 (DOI: 10.1080/ 17451000.2014.894244).

Martinho F. (2012) *Residency and behavioural patterns of coastal bottlenose dolphins (Tursiops truncatus) in the Arrábida and Tróia shores (Portugal).* MSc thesis, Faculdade de Ciências da Universidade de Lisboa, Portugal. 80pp.

Mead, J.C. and Potter, C.W. (1995) Recognizing two populations of the bottlenose dolphin (*Tursiops truncatus*) of the Atlantic coast of North America - morphologic and ecologic considerations. *IBI Reports* (International Marine Biological Research Institute, Kamogawa, Japan), 5: 31**–**44.

Mirimin, L., Miller, R., Dillane, E., Berrow, S.D., Ingram, S., Cross, T.F., and Rogan, E. (2011) Fine-scale population genetic structuring of bottlenose dolphins in Irish coastal waters. *Animal Conservation*, 14: 342-353.

Mohr, E. (1935) Historisch-zoologische Walfischstudien. *Nordelbingen*, 11: 335-393.

Möller, L.M, and Beheregaray, L.B. (2004) Genetic evidence for sex-biased dispersal in resident bottlenose dolphins (*Tursiops aduncus*). *Molecular Ecology*, 13: 1607–1612.

Natoli, A., Peddemors, V.M., and Hoelzel, A.R. (2004) Population structure and speciation in the genus Tursiops based on microsatellite and mitochondrial DNA analyses. *Journal of Evolutionary Biology*, 17: 363**–**375.

Natoli, A., Birkun, A., Aguilar, A., López, A., and Hoelzel, A.R. (2005) Habitat structure and the dispersal of male and female bottlenose dolphins (*Tursiops truncatus*). *Proceedings of the Royal Society B - Biological Sciences*, 272: 1217–1226.

Nichols, C., Herman, J., Gaggiotti, O.E., Dobney, K.M., Parsons, K. and Hoelzel, A.R. (2007) Genetic isolation of a now extinct population of bottlenose dolphins (*Tursiops truncatus*). *Proceedings of the Royal Society B*, 274. DOI: 10.1098/rspb.2007.0176.

NOAA (2016) Guidelines for preparing stock assessment reports pursuant on the 1994 amendments to the MMPA. National Marine Fisheries Service Instruction 02-204-01, February 22, 2016. 24pp.

Normann, E.B., Dussan-Duque, S., and Evans, P.G.H. (2015) *Bottlenose dolphins in Wales: Systematic mark-recapture surveys in Welsh waters*. Natural Resources Wales Evidence Report Series No. 85. Natural Resources Wales, Bangor. 83pp.

Nykanen, M., Ingram, S.D., and Rogan, E. (2015) *West coast dolphins (Tursiops truncatus): abundance, distribution, ranging patterns and habitat use*. Report to the National Parks and Wildlife Service. 33pp.

O'Brien, J.M., Berrow, S.D., Ryan, C., McGrath, D., O'Connor, I., Pesante, G., Burrows, G., Massett, N., Klötzer, V., and Whooley, P. (2009) A note on long-distance matches of bottlenose dolphins *(Tursiops truncatus*) around the Irish coast using photo-identification. *Journal of Cetacean Research and Management,* 11: 69–74.

Ó Cadhla, O., Englund, A., Philpott, E., Mackey, M. and Ingram, S.N. (2003) *Marine mammal monitoring in the waters of Broadhaven Bay and Northwest Mayo: 2001-2002.* Report to Enterprise Energy Ireland Ltd, 74pp.

Oudejans, M., Ingram, S., Englund, A., Visser, F. and Rogan, E. (2010) Bottlenose dolphins in Connemara and Mayo 2008-2009. Movement patterns between two coastal areas in the west of Ireland. Report to the National Parks and Wildlife Service Department of the Environment, Heritage and Local Government.

Oudejans, M., Visser, F., Englund, A., Rogan, E., and Ingram, S.N. (2015) Evidence for Distinct Coastal and Offshore Communities of Bottlenose Dolphins in the North East Atlantic. PLOS One, DOI:10.1371/journal.pone.0122668.

Palsbøll, P.J., Bérubé, M., and Allendorf F.W. (2007) Identification of management units using population genetic data. *Trends in Ecology & Evolution*, 22: 11-16.

Perrin, W.F., Thieleking, J.L., Walker, W.A., Archer, F.I., and Robertson, K.M. (2011) Common bottlenose dolphins (*Tursiops truncatus*) in California waters: cranial differentiation of coastal and offshore ecotypes. *Marine Mammal Science*, 27: 769–792.

Pesante, G., Evans, P.G.H., Baines, M.E., and McMath, M. (2008). *Abundance and Life History Parameters of Bottlenose Dolphin in Cardigan Bay: Monitoring 2005-2007.* CCW Marine Monitoring Report No. 61. 75pp.

Pirotta, E., Merchant, N.D., Thompson, P.M., Barton, T.R., and Lusseau, D. (2015) Quantifying the effect of boat disturbance on bottlenose dolphin foraging activity. *Biological Conservation*, 181: 82-89.

Pirotta, E., Thompson, P.M., Cheney, B., Donovan, C.R., and Lusseau, D. (2014) Estimating spatial, temporal and individual variability in dolphin cumulative exposure to boat traffic using spatially explicit capture–recapture methods. *Animal Conservation*, DOI: 10.1111/acv.12132.

Portillo, C., Jiménez-Torres, C., Pérez, S.,Verborgh, P., Gauffier, P., Esteban, R., Giménez, J.,Santos-Vegas, M., Cazalla, E., and de Stephanis, R. (2011) Survival rate, abundance and residency of bottlenose dolphins in the Strait of Gibraltar. Poster. 25th Conference of the European Cetacean Society, 21-23 March 2011, Cádiz. Spain.

Quérouil, S., Silva, M.A., Freitas, L., Prieto, R., Magalhães, S., Dinis, A., Alves, F., Matos, J.A., Mendonça, D., Hammond, P., and Santos, R.S. (2007) High gene flow in oceanic bottlenose dolphins (*Tursiops truncatus*) of the North Atlantic. *Conservation Genetics*, 8: 1405–1419.

Reid, J.B., Evans, P.G.H., & Northridge, S.P., (2003), Atlas of Cetacean distribution in north-west European waters, 76 pages, colour photos, maps. Paperback, ISBN 1 86107 550 2.

Richards, V.P., Greig, T.W., Fair, P.A., McCulloch, S.D., Politz, P., Natoli, A., Driscoll, C.A. Hoelzel, A.R., David, V., Bossart, G.D., and Lopez, J.V. (2013) Patterns of population structure for inshore bottlenose dolphins along the eastern United States. *Journal of Heredity,* 104(6): 765-778.

Rogan, E., Nykanen, M. Gkaragkouni, M., and Ingram, S.N. (2015) *Bottlenose dolphin survey in the lower River Shannon SAC, 2015.* Report to the National Parks and Wildlife Service. 21pp.

Roman, J., and Palumbi, S.R. (2003) Whales before Whaling in the North Atlantic. *Science*, 301: 508-510.

Rosel, P.E., Hansen, L., and Hohn, A.A. (2009) Restricted dispersal in a continuously distributed marine species: common bottlenose dolphins *Tursiops truncatus* in coastal waters of the western North Atlantic. *Molecular Ecology*, 18: 5030–5045.

Ruegg, K., Rosenbaum, H.C., Anderson, E.C., Engle, M., Rothschild, A., Baker, C.S. and Palumbi, S.R. (2013) Long-term population size of the North Atlantic humpback whale within the context of worldwide population structure. *Conservation Genetics*, 14: 103-114.

Santos, J., Araújo, H., Ferreira, M., Henriques, A., Miodonski, J., Monteiro, S., Oliveira, I., Rodrigues, P., Duro, G., Oliveira, F., Pinto, N., Sequeira, M., Eira, C. and Vingada, J. (2012) *Chapter I: Baseline estimates of abundance and distribution of target species*. Annex to the Midterm Report of Project LIFE MarPro NAT/PT/00038.

Teixeira, A.M.A.P. (1979) Marine mammals of the Portuguese coast. *Z. Säugetierkunde*, 44: 221-238.

Toth, J.L. Hohn, A.A. Able, K.W., and Gorgone, A.M. (2012) Defining bottlenose dolphin (*Tursiops truncatus*) stocks based on environmental, physical, and behavioral characteristics. *Marine Mammal Science*, 28(3): 461-478.

Tregenza, N.J.C. (1992) Fifty years of cetacean sightings from the Cornish coast, SW England. *Biological Conservation,* 59: 65-70.

Van Bressem, M.F., Gaspar, R. and Aznar, F.J. (2003) Epidemiology of tattoo skin disease in bottlenose dolphins *Tursiops truncatus* from the Sado estuary, Portugal. Diseases of Aquatic Organisms, 56(2): 171-179.

Van Geel, N. (2016) *Predator movements in complex geography: Spatial distribution and temporal occurrence of low-density bottlenose dolphin communities off western Scotland.* PhD thesis, University of Aberdeen. 459pp.

Vázquez, J.A., Cañadas, A., Martínez-Cedeira, J., López, A., Tejedor, M., Gauffier, P., Gazo, M., and Brotons, J.M. (2014) *Docuemento técnico sobre la incidencia de la captura accidental de especies de cetáceo amenazadas en artes de pesca*. Informe realizado para el Ministerio de Agricultura, Alimentacíon y Medio Ambiente. 95pp.

Vélez, M. (2014) *Interacciones entre cetáceos, Tortugas y pesquerías en el Golfo de Cádiz*. MSc Thesis, University of Cádiz. 65pp.

Verwey, J. (1975) The cetaceans *Phocoena phocoena* and *Tursiops truncatus* in the Marsdiep area (Dutch Waddensea) in the years 1931-1973. *Nederlands Instituut voor Onderzoek der Zee,* 17a, b: 1-153.

Walker, W.A. (1981) Geographic variation in morphology and biology of bottlenose dolphins (Tursiops) in the eastern North Pacific. NOAA/NMFS Administrative Report LJ-81-0003c, 21pp.

Waring, G.T., Josephson, E., Maze-Foley, K., and Rosel, P.E. (2016) US Atlantic and Gulf of Mexico Marine Mammal Stock assessments – 2015. NOAA Technical Memorandum NMFS-NE-238. 512pp.

Weir, C.R. and Stockin, K.A. (2001) *The occurrence and distribution of bottlenose dolphins (Tursiops truncates) and other cetacean species in the coastal waters of Aberdeenshire, Scotland*. Sea Watch Foundation, Aberdeen. 68pp.

Williams, A.D., Williams, R., Heimlich-Boran., J.R., Evans, P.G.H., Tregenza, N.J.C., Ridoux, V. Liret, C., and Savage, S. (1997) A preliminary report on an investigation into bottlenose dolphins (Tursiops truncatus) of the English Channel: a collaborative approach. European Research on Cetaceans, 10: 217-220.

Wilson, B., Reid, R.J., Grellier, K., Thompson, P.M., and Hammond, P.S. (2004) Considering the temporal when managing the spatial: a population range expansion impacts protected areas-based management for bottlenose dolphins. *Animal Conservation,* 7: 331-338.

Wilson, B., Thompson, P.M., and Hammond, P.S. (1997) Habitat use by bottlenose dolphins: seasonal distribution and stratified movement patterns in the Moray Firth, Scotland. *Journal of Applied Ecology*, 3: 1365-1374.

Wiszniewski, J., Allen, S.J., and Moeller, L.M. (2009) Social cohesion in a hierarchically structured embayment population of Indo-Pacific bottlenose dolphins. *Animal Behaviour*, 77: 1449–1457.

Wiszniewski, J., Beheregaray, L.B, Allen, S.J., and Möller, L.M. (2010) Environmental and social influences on the genetic structure of bottlenose dolphins (*Tursiops aduncus*) in Southeastern Australia. *Conservation Genetics*. 11: 1405–1419.

1.2 Short-beaked common dolphin (*Delphinus delphis*) in Europe

Step 1.1 - Biology of the species

The short-beaked common dolphin (*Delphinus delphis*) is one of the smallest of the true dolphins (family Delphinidae). It is an active, fast-moving species, frequently bow-riding boats and jumping clear of the water. Two species of common dolphin are currently recognised in the North Atlantic – the short-beaked form, *D. delphis*, and the long-beaked form, *D. capensis,* but only the former has been recorded in European seas. It is a social species, usually travelling in schools of 6-15 individuals, but can form aggregations numbering hundreds or even low thousands, associated with feeding or large-scale movements. There is evidence for some segregation by age and sex. Food-herding behaviour is frequently observed with apparent cooperation between school members. Mating and calving are seasonal, largely between May and September. Males reach sexual maturity around 11-12 years, females around 9-10 years of age. On reaching sexual maturity, females give birth to a single calf, every 3-4 years. Maximum age reported in the North-east Atlantic is 28 years. Generation length has been estimated at 14.1 years (14.8 years in populations at a stable state). Common dolphins appear to be opportunistic feeders, but most common prey recorded in the region are epi- and mesopelagic fish including horse mackerel, mackerel, Norway pout, sardines, anchovy, whiting, scad, sprat, sandeel, and blue whiting, as well as small cephalopods. More detailed accounts of the biology of the species can be found in Evans (1994), Heyning & Perrin (1994), Murphy *et al.* (2008, 2013), and Perrin (2009).

Step 1.2 - Spatial scale of functioning

The short-beaked common dolphin has a worldwide distribution in oceanic and shelf-edge waters of tropical, subtropical and temperate seas, occurring in both hemispheres. It is abundant and widely distributed in the eastern North Atlantic, mainly occurring in deeper waters from Macaronesia and north-west Africa north to approximately 65ºN latitude (although rare north of 62º N), west of Norway and the Faroe Islands (Reid *et al.* 2003, Murphy *et al.* 2008). It occurs westwards at least to the mid-Atlantic ridge (Doksaeter *et al.* 2008, Cañadas *et al.* 2009), and eastwards it enters the western Mediterranean, with a distinct isolated population in the Black Sea (Bearzi *et al.* 2003) (Figure 1.2.1). Sightings are rare in the eastern section of the English Channel and the southern North Sea (Evans *et al.* 2003, Camphuysen & Peet 2006). In the offshore North Atlantic it seems to favour waters over 15º C and shelf edge features at depths of 400-1,000m between 49º-55º N especially between 20º-30º W (Forcada *et al.* 1990, Cañadas *et al.* 2009).

Figure 1.2.1 *North Atlantic distribution of short-beaked common dolphin (depicting those areas where the species is thought to regularly occur)*

On the UK continental shelf, the species is common in the western half of the English Channel and the southern Irish Sea, and further north in the Sea of Hebrides and southern part of the Minch (Evans *et al.* 2003, Reid *et al.* 2003). It is also common south and west of Ireland. In some years, the species occurs further north and east in shelf seas - in the northern Hebrides, around Shetland and Orkney, and in the northern North Sea. It is generally rare in the central and southern North Sea and eastern portion of the English Channel, but is abundant in the Bay of Biscay (Evans *et al.* 2003, Reid *et al.* 2003, Kiszka *et al.* 2007, Laran *et al.* 2017). Range shifts may be related to changing oceanographic conditions (warmer sea temperatures and extensions of the range of potential prey species like anchovy and sardine). Since the 1990s, the species has become regular in the northern North Sea and even entered the Baltic (Evans *et al.* 2003, Murphy *et al.* 2008, Kinze *et al.* 2010).

In shelf waters off the west coasts of Ireland and Scotland, and in the Irish Sea, common dolphin abundance tends to be greatest in the summer months (Evans *et al.* 2003, Berrow *et al.* 2010), whereas further south, the western English Channel and Bay of Biscay remain important winter habitats for the species, which is abundant there between December and March (Brereton *et al.* 2005, De Boer *et al.* 2008), although abundance in the Bay of Biscay is greatest in the summer (Laran *et al.* 2016). The species also occurs all along the coast of Portugal and southwest Spain.

Although common dolphins can be found over large parts of the eastern North Atlantic and appear to form one large panmictic population (Mirimin *et al.* 2009, Amaral *et al.* 2012, Moura *et al.* 2013), there is also some suggestion for the existence of separate populations, e.g. one occurring primarily offshore and the other over the continental shelf, although this needs further clarification (Caurant *et al.* 2009, Evans & Teilmann 2009). Genetic (mtDNA) studies indicate a significant level of divergence between Mediterranean (Alboràn Sea) and Atlantic populations (Natoli *et al.* 2008), although with directional estimates of gene flow suggesting some movement of females out of the Mediterranean (possibly due to oceanographic features such as the Almería-Orán thermohaline front). Differences in contaminant levels between dolphins from the Alboràn Sea and Atlantic Ocean also suggest a certain degree of isolation (Borrell *et al.* 2001).

Although common dolphins are relatively abundant in the westernmost portion of the Mediterranean basin - the Alboràn Sea, the species is rare or completely absent from large parts of the Mediterranean (Bearzi *et al.* 2003). Mediterranean regions where common dolphins have apparently vanished include the Adriatic Sea, Balearic Sea, Provençal basin, and Ligurian Sea, the Saronic Gulf, and the Dodekanese (Frantzis *et al.* 2003). There are only sparse records with possibly isolated groups present around Sardinia and Corsica (Bearzi *et al.* 2003), and seasonally in the south-eastern Tyrrhenian Sea off the island of Ischia (Mussi *et al.* 2002). The species is present in the Sicily Channel and around Malta (Vella 2002), in portions of the eastern Ionian Sea, particularly around the island of Kalamos (Politi & Bearzi 2001), and in the Gulf of Corinth (Frantzis & Herzing 2002). Genetic studies show differentiation between the western and eastern Mediterranean populations, and suggest that a population bottleneck likely occurred as a result of a sharp population decline in the recent past (Natoli *et al.* 2008, Moura *et al.* 2013).

At the eastern end of the Mediterranean, there is little indication of movement by common dolphins through the narrow Dardanelles Strait between the Aegean and the Marmara and Black Seas, where common dolphins are known to occur (Öztürk & Öztürk 1997, Frantzis *et al.* 2003). A preliminary study of skull morphometrics (Amaha 1994) suggested differences between Black Sea and Mediterranean common dolphins. However, a genetic comparison of relatively small samples (8 Black Sea, 20 central Mediterranean) revealed no significant differences (Natoli *et al.* 2008). Clearly, further work based on larger samples is needed to assess and characterize the relationship between Black Sea and Mediterranean common dolphins. Some genetic exchange may occur in portions of the Aegean Sea where favourable habitat still exists (e.g., in the Thracian Sea; Frantzis *et al.* 2003) but what remains between the Aegean and Alboràn sectors of the Mediterranean seems to be only isolated, remnant groups (possibly indicative of further population substructure). The Black Sea population therefore remains classified as a separate subspecies *D. delphinus ponticus*.

Conclusions

Current knowledge indicates that the short-beaked common dolphin in the Atlantic and North Sea should be listed under category MR4, a population with individuals having inherently large home ranges forming one mixing population, or in category MR1 as it has a clearly sustainable meta-population above Member State level and is not generally considered to be a fully migratory species. The populations in the Alboràn Sea (western Mediterranean) and Aegean Sea (eastern Mediterranean) are geographically isolated and genetically distinct, as may be the one in the Black Sea. Since these fragmented populations nevertheless exist above Member State level, they would most closely fit category S4 as well.

Step 1.3 - Historical perspective: what happened to the species?

There is little historical information from the European Atlantic to indicate trends in common dolphin abundance. The species has occurred in variable frequency in the North Sea with no particular trend (Evans *et al.* 2003, Reid *et al.* 2003, Camphuysen & Peet 2008), whereas it has been only infrequent in the Danish Belt Seas and Baltic (Kinze 1995). Once one of the most abundant species in the Mediterranean Sea, the common dolphin has experienced a generalized and major decrease in this region since the middle of the twentieth century (Bearzi *et al.* 2003). Dramatic declines were recorded in portions of the central Mediterranean, particularly in the northern Adriatic Sea and in the eastern Ionian Sea (Bearzi *et al.* 2004, 2006).

In the Black Sea, common dolphin populations collapsed by the mid 1960s following many years of overexploitation, which involved the killing of several hundreds of thousands of common dolphins in the mid-20th century (IWC 1983, Birkun 2006, 2008). Commercial killing of Black Sea common dolphins, as well as other Black Sea cetaceans, was banned in 1966 in the former USSR, Bulgaria and Romania, and in 1983 in Turkey. Since then, the population has shown little sign of recovery (Birkun 2006, 2008). It was calculated that between 1962 and 1966, the cetacean fishery in the USSR and Bulgaria landed 121,395 common dolphins, while during the preceding 31 years (1931-61) a further 1,449,304 had been landed mainly by the USSR (Zemsky 1996). Between 1976 and 1981, common dolphin was believed to account for 15-16% (or 37,500-40,000 individuals) of the Turkish catch, estimated for that period as 250,000 animals of all three species (common dolphin, harbour porpoise, and bottlenose dolphin) (IWC 1983).

In the European Atlantic, the main threat to common dolphins is probably bycatch, involving French, Spanish, Portuguese, Irish and UK pelagic trawl fisheries targeting a range of fish including albacore tuna, sea bass, blue whiting, hake, horse mackerel, sardine or anchovy (Tregenza *et al.* 1997, Tregenza & Collet 1998, Morizur *et al.* 1996, 1999, 2014, ICES 2016). Annual bycatch levels have been estimated in the hundreds or low thousands, from independent observer programmes, although these have not comprehensively assessed all fisheries. During the 1990s, the tuna driftnet fishery also caught large numbers of common dolphins until a ban was introduced in 2002 (Goujon *et al.* 1993, Goujon 1996, Rogan & Mackey 2007). Bycatches are also known to occur in gill nets, tangle nets and possibly other fisheries (Tregenza et al 1997, Cosgrove & Browne 2007). From post mortem examinations of 537 common dolphins stranded in the UK between 1991 and 2010, 51% were diagnosed as by-catch, 18% live-stranded, 7% from infectious disease, and 4% died from starvation (Deaville & Jepson 2011).

In the Mediterranean, incidental mortality in fishing gear, particularly driftnets, is thought to have contributed to the decline of common dolphins, although reduced availability of prey caused by overfishing and habitat degradation, contamination by persistent chemicals resulting in immunosuppression and reproductive impairment, and environmental changes such as increased water temperatures affecting ecosystem dynamics are also thought to have played a role (Bearzi *et al.* 2003, 2006). The Moroccan driftnet fishing fleet has been estimated to have an annual bycatch of c. 12,000-15,000 dolphins around the Strait of Gibraltar (Tudela *et al.* 2004).

In the Black Sea, besides the very large numbers of common dolphins hunted in the last century until bans came into force in the 1960s-80s, reduced prey availability (e.g. of anchovy and sprat) has been considered a major threat common dolphins (Bushuyev 2000). This was thought to contribute to the two mass mortality events that killed large numbers of common dolphins in 1990 and 1994

(Krivokhizhin & Birkun 1999), in the case of the latter linked to increased susceptibility to morbillivirus infection. Prey reduction is thought to have been caused by a combination of overfishing, eutrophication and the explosive increase of the introduced ctenophore *Mnemiopsis leidyi* (Birkun 2006). Prey depletion caused by overfishing was also considered as a main cause for the decline of common dolphins in the eastern Ionian Sea (Bearzi *et al.* 2006).

Step 1.4 - Analysis of distribution and trends

There have been several abundance surveys of common dolphin in various parts of the eastern North Atlantic though none spanning the entire region (e.g. MICA survey in 1993 - Goujon *et al.* 1993; ATLANCET aerial survey in 2001 - Ridoux *et al.* 2003; MARPRO surveys in 2007-12). In NW Europe, the SCANS survey (July 1994), covering an area from the Celtic shelf to c. 11°W and 48°S, produced an estimate of 75,449 individuals (CV=0.67; 95% CI: 23,900–248,900) (Hammond *et al.* 2002). However, this survey did not use a double-platform method, nor correct for animals missed on the track line, nor, perhaps most importantly, responsive movement – common dolphins notoriously respond positively to the presence of vessels that they can bow ride. In 2005, SCANS-II re-surveyed the same area as SCANS, but extended this to include also the Irish Sea, the waters off western and Northern Ireland, west Scotland, and continental shelf waters off France, Spain and Portugal. The total summer abundance for those Northeast Atlantic shelf waters was c. 56,221 (CV=0.23; 95% CI: 35,750–88,400) (Hammond et al 2013). This was supplemented by the CODA offshore survey conducted in July 2007 along the shelf edge of the ASCOBANS Agreement Area, which estimated a total abundance of 116,709 (CV=0.34; 95% CI: 61,400–221,800) (CODA 2009). These surveys corrected for animals missed along the track-line but not for responsive movement. The only analysis in the region that has not only corrected for animals missed along the track-line but also for responsive movement was for the western block of the 1995 NASS survey that covers part of the central North Atlantic (52º–57.5º N, 18º–28º W), and resulted in an estimate of 273,000 common dolphins (CV=0.26; 95% CI: 153,000–435,000) (Cañadas *et al.* 2009).

In July 2016, SCANS-III survey was undertaken and yielded an abundance estimate of 467,673 (CV=0.26; 95% CI: 281,100-778,000). This estimate excludes results from western and southern Ireland which are not yet available. The most striking difference is the much larger estimates of this species and striped dolphin (together totalling almost a million animals) in 2016 compared to 2005/07, nevertheless consistent with results from the French SAMM surveys in French waters of the Bay of Biscay and the Channel in summer 2012, which totalled almost 700,000 common and striped dolphins (Laran *et al.* 2017). Notwithstanding this, the lack of estimates for Irish waters in 2016 precludes any robust comparisons of estimates of abundance between 2005/07 and 2016 for the whole area.

In the western Mediterranean, abundance has been estimated at 19,400 (95%CI: 15,300-22,800) in the northern Alborán Sea between 2000 and 2004 (Cañadas & Hammond 2008). The size of the eastern Mediterranean population has not been estimated.

The population size in the Black Sea is unknown. Line transect surveys have been conducted recently to estimate common dolphin abundance in a few parts of the range. The survey areas are small relative to the total range of the subspecies. Results suggest that current population size is at least several tens of thousands, and possibly 100,000 or more (Birkun 2006).

Conclusions

The common dolphin was much more widespread in the Mediterranean in the middle of the last century, and in the Black Sea a crash in numbers was documented between the 1940s and the 1960s. Declines in the Mediterranean were thought to be caused by a combination of heavy bycatch and prey depletion. One obvious cause of the declines in the Black Sea was the excessive hunting of the species by several countries. That came to an end in the 1980s but since then the Black Sea population has shown little sign of recovery, probably because it is still experiencing prey depletion from overfishing and habitat degradation from pollution. If these human pressures were reduced, there is scope for recovery of depleted populations.

Common dolphins in the eastern North Atlantic appear to be in a healthier state although they continue to suffer fisheries bycatch in particular.

As with other cetacean species, long-terms trends in common dolphin numbers are generally lacking. There are no data on population sizes for the species in Europe prior to human impacts, so it is not possible to set a historical baseline. However, there is good evidence to indicate major declines over a large part of the Mediterranean and in the Black Sea. The three (1994, 2005/07, 2016) wide-scale July surveys in the European Atlantic are difficult to interpret because coverage has varied between surveys, and none has encompassed the entire eastern North Atlantic range. Nevertheless, they do not suggest a decline, and the variation observed may simply reflect differential movement between regions.

Step 2.1 - FRP assessment

In the case of common dolphins in the European Atlantic, for which there is no historic evidence before 1992 when the Habitats Directive came into force but no obvious contraction in range or declines in orders of magnitude of abundance, it probably makes sense to treat FRP as equivalent to CV. For the Mediterranean and Black Sea, that is clearly not the case, the former having experienced a major contraction in range, and the latter in population size. In those regions, FRP should greatly exceed CV.

Whole genomic analysis using restriction site associated DNA sequencing is recommended for insights into the extent to which present day populations have experienced contractions in size and loss of genetic diversity. Genetic analysis enables one to estimate the effective population size (Ne) for management units prior to major human impacts.

Conclusions

Short-beaked common dolphin and its close relative long-beaked common dolphin are amongst the most abundant of dolphin species globally. The order of magnitude of FRP in European seas is likely to be in the range of one to two million animals, but for separate populations, it will be correspondingly smaller. The major declines that have occurred historically in the Mediterranean and Black Sea could be reversed if conditions allowed.

FRP = CV (European Atlantic, MATL and MMAC)
FRP > CV (Mediterranean, MMED)
FRP >> CV (Black Sea, MBLS)

Step 2.2 - FRR assessment

Common dolphins are wide ranging in the North Atlantic both on and off the shelf. The species also occurs in the Mediterranean and Black Sea, although in the case of the former, its range appears to be significantly reduced. Its current range at least in the Mediterranean could therefore likely expand if suitable conditions prevailed.

Conclusions

FRR = CV (European Atlantic, MATL and MMAC)
FRR >> CV (Mediterranean, MMED)
FRP > CV (Black Sea, MBLS)

References

Amaha, A. (1994) Geographic variation of the Common Dolphin, *Delphinus delphis* (Odontoceti: Delphinidae). Ph.D. thesis, Tokyo University of Fisheries. 211pp.

Amaral, A.R., Luciano B. Beheregaray, L.B., Bilgmann, K., Dmitri Boutov, D., Freitas, L., Robertson, K.M., Sequeira, M., Stockin, K.A., Coelho, M.M., Luciana, M., and Møller, L.M. (2012) Seascape genetics of a globally distributed, highly mobile marine mammal: the short-beaked common dolphin (Genus *Delphinus*). *PLoS One*, 7(2). http://dx.doi.org/10.1371/journal.pone.0031482.

Bearzi, G., Reeves, R.R., Notarbartolo-di-Sciara, G., Politi, E., Cañadas, A., Frantzis, A., and Mussi, B. (2003) Ecology, status and conservation of short-beaked common dolphins *Delphinus delphis* in the Mediterranean Sea. *Mammal Review*, 33: 224-252.

Bearzi, G., Holcer, D., and Notarbartolo Di Sciara, G. (2004) The role of historical dolphin takes and habitat degradation in shaping the present status of northern Adriatic cetaceans. *Aquatic Conservation of Marine and Freshwater Ecosystems* 14: 363-379.

Bearzi, G., Politi, E., Agazzi, S., and Azzellino, A. (2006) Prey depletion caused by overfishing and the decline of marine megafauna in eastern Ionian Sea coastal waters (central Mediterranean). *Biological Conservation* 127(4): 373-382.

Birkun Jr., A.A. (2006) Short-beaked common dolphin (*Delphinus delphis ponticus*): Black Sea subspecies. In: R. R. Reeves and G. Notarbartolo di Sciara (eds), *The status and distribution of cetaceans in the Black Sea and Mediterranean Sea*, pp. 16-22. IUCN Centre for Mediterranean Cooperation, Malaga, Spain.

Birkun Jr., A.A. (2008) *Delphinus delphis ssp. ponticus. The IUCN Red List of Threatened Species 2008*: e.T133729A3875256. http://dx.doi.org/10.2305/IUCN.UK.2008.RLTS.T133729A3875256.en

Brereton, T., Williams, A., and Martin, C. (2005) Ecology and status of the common dolphin *Delphinus delphis* in the English Channel and Bay of Biscay 1995–2002. Pp. 15-22. In Proceedings of the Workshop on Common Dolphins: Current Research, Threats and Issues, Kolmården, Sweden 1st April, 2004, (eds. K. Stockin *et al.*). European Cetacean Society Special Issue.

Bushuyev, S.G. (2000) Depletion of forage reserve as a factor limiting population size of Black Sea dolphins. *Ecological Safety of Coastal and Shelf Areas and a Composite Utilization of Shelf Resources*, pp. 437-452. Proc. Marine Hydrophysical Institute, Sevastopol.

Camphuysen, C.J. and Peet, G. (2006) *Whales and dolphins in the North Sea*. Fontaine Uitgevers, Kortenhoef, The Netherlands.

Cañadas, A. and Hammond, P.S. (2008) Abundance and habitat preferences of the short-beaked common dolphins *Delphinus delphis* in the western Mediterranean: implications for conservation. *Endangered Species Research*, 4: 309-331.

Cañadas, A., Desportes, G., Borchers, D., and Donovan, G. (2009) A short review of the distribution of short-beaked common dolphin (*Delphinus delphis*) in the central and eastern North Atlantic with an abundance estimate for part of this area. *NAMMCO Scientific Publications* Vol. 7: 201-220.

Caurant, F., Chouvelon, T., Lahaye, V., Mendez-Fernandez P., Rogan E., Spitz J., and Ridoux V. (2009) The use of ecological tracers for discriminating dolphin population structure: the case of the short-beaked common dolphin *Delphinus delphis* in European Atlantic waters. IWC SC/61/SM34. 11pp.

Cosgrove, R. and Browne, D. (2007) Cetacean by-catch rates in Irish gillnet fisheries in the Celtic Sea. Marine Technical Report, June 2007. Dun Laoghaire, Co. Dublin: Bord Iascaigh Mhara.

De Boer, M.N., Leaper, R., Keith, S., and Simmonds, M.P. (2008) Winter abundance estimates for the common dolphin (Delphinus delphis) in the western approaches of the English Channel and the effect of responsive movement. *Journal of Marine Animals and Their Ecology*, 1(1): 15-21.

Doksaeter, L., Olsen, E., Nottestad, L., and Ferno, A. (2008) Distribution and feeding ecology of dolphins along the Mid-Atlantic Ridge between Iceland and the Azores. *Deep Sea Research Part II: Topical Studies in Oceanography*, 55: 243-253.

Evans, P.G.H., Anderwald, P., and Baines, M.E. (2003) *UK Cetacean Status Review.* Report to English Nature and Countryside Council for Wales. Sea Watch Foundation, Oxford. 160pp.

Evans, W.E. (1994) Common Dolphin, White-bellied Porpoise *Delphinus delphis* Linnaeus, 1758. In: *Handbook of Marine Mammals. Volume 5: The First Book of Dolphins*. Academic Press, London, pp. 191-224.

Forcada, J., Aguilar, A., Evans, P.G.H., and Perrin, W.F. (1990) Distribution of common and striped dolphins in the temperate waters of the eastern North Atlantic. *European Research on Cetaceans*, 4: 64-66.

Frantzis, A. and Herzing, D.L. 2002. Mixed species associations of Striped Dolphin (*Stenella coerule oalba*), Short-beaked Common Dolphin (*Delphinus delphis*) and Risso's Dolphin (*Grampus griseus*), in the Gulf of Corinth (Greece, Mediterranean Sea). *Aquatic Mammals* 28(2): 188-197.

Frantzis, A., Alexiadou, P., Paximadis, G., Politi, E., Gannier, A., and Corsini-Foka, M. (2003) Current knowledge on the cetacean fauna of the Greek Seas. *Journal of Cetacean Research and Management,* 5(2): 219-232.

Goujon, M. (1996) *Captures accidentelles du let maillant derivant et dynamique des populations de dauphins au large du Golfe de Gascogne*. PhD thesis, école Nationale Superieure agronomique de Rennes, France.

Goujon, M., Antoine, L., Collet, A., and Fifas S. (1993) *Approche de l'mpact de la ecologique de la pecherie thoniere au filet maillant derivant en Atlantique nord-est*. Rapport internese al Direction des Resources Vivantes de I'IFREMER., Ifremer, Centre de Brest, BP 70, 29280 Plouzane, France.

Hammond, P.S., Berggren, P., Benke, H., Borchers, D.L., Collet, A., Heide-Jørgensen, M.P., Heimlich, S., Hiby, A.R., Leopold, M.F., and Øien, N. (2002) Abundance of harbour porpoise and other cetaceans in the North Sea and adjacent waters. *Journal of Applied Ecology*, 39: 361-376.

Hammond, P.S., Macleod, K., Berggren, P., Borchers, D.L., Burt, M.L., Cañadas, A., Desportes, G., Donovan, G.P., Gilles, A., Gillespie, D., Gordon, J., Hiby, L., Kuklik, I., Leaper, R., Lehnert, K., Leopold, M., Lovell, P., Øien, N., Paxton, C.G.M., Ridoux, V., Rogan, E., Samarra, F., Scheidat, M., Sequeira, M., Siebert, U., Skov, H., Swift, R., Tasker, M.L., Teilmann, J., Van Canneyt, O., and Vázquez, J.A. (2013) Cetacean abundance and distribution in European Atlantic shelf waters to inform conservation and management. *Biological Conservation,* 164**:** 107–122.

Heyning, J.E. and Perrin, W.F. (1994) Evidence for two species of common dolphins (genus *Delphinus*) from the eastern North Pacific. *Contributions in Science* (Los Angeles County Museum) 442: 1–35.

IWC (1983) Report of the Sub-Committee on Small Cetaceans. *Report of The International Whaling Commission*, 33: 152-170.

Kinze, C.C. (1995) Danish whale records 1575-1991 (Mammalia, Cetacea). *Steenstrupia*, 21: 155-196.

Krivokhizhin, S. V. and Birkun Jr., A. A. (1999) Strandings of cetaceans along the coasts of Crimean Peninsula in 1989-1996. *European Research on Cetaceans,* 12: 59-62.

Laran, S., Authier, M., Blanck, A., Doremus, G., Falchetto, H., Monestiez, P., Pettex, E., Stephan, E., Van Canneyt, O., and Ridoux, V. (2017) Seasonal distribution and abundance of cetaceans within French waters: Part II: The Bay of Biscay and the English Channel. *Deep-Sea Research II*, 14: 31-40.

Mirimin, L., Westgate, A., Rogan, E., Rosel, P., Read A., Coughlan, J., and Cross, T. (2009) Population structure of short-beaked common dolphins (*Delphinus delphis*) in the North Atlantic Ocean as revealed by mitochondrial and nuclear genetic markers. *Marine Biology*, 156: 821–834.

Morizur, Y., Pouvreau, S., and Guenole, A. (1996) *Les rejets dans la pêche artisanale française de Manche occidentale.* Editions, IFREMER, France. 127pp.

Morizur, Y., Berrow, S.D, Tregenza, N.J.C., Couperus, A.S., and Pouvreau, S. (1999) Incidental catches of marine mammals in pelagic trawl fisheries of the north-east Atlantic. *Fisheries Research*, 41(3): 297-307.

Morizur, Y., Gaudou, O., and Demaneche, S. (2014) Analyse des captures accidentelles de mammifères marins dans les pêcheries françaises aux filets fixes. http://archimer.ifremer.fr/doc/00209/32016/

Moura, A., Natoli, A., Rogan, E., and Hoelzel, A.R. (2013) Atypical panmixia in a European dolphin species (*Delphinus delphis*): Implications for the evolution of diversity across oceanic boundaries***.*** *Journal of Evolutionary Biolog*y 26(1):

Murphy, S., Evans, P.G.H., and Collet, A. (2008) Common dolphin *Delphinus delphis*. Pp. 719-724. In: *Mammals of The British Isle*s (Eds. S. Harris & D.W. Yalden). Handbook, 4th Edition. The Mammal Society, Southampton. 800pp.

Murphy, S., Pinn, E.H., and Jepson, P.D. (2013) The short-beaked common dolphin (*Delphinus delphis*) in the north-eastern Atlantic: distribution, ecology, management and conservation status. *Oceanography and Marine Biology: An Annual Review*, 51: 193-280.

Mussi, B., Miragliuolo, A. and Bearzi, G. (2002) Short-beaked common dolphins around the island of Ischia, Italy (Southern Tyrrhenian Sea). European Research on Cetaceans 16. 132-134.

Natoli, A., Cañadas, A., Vaquero, C., Politi, E., Fernandez-Navarro, P., and Hoelzel, A.R. (2008) Conservation genetics of the short-beaked common dolphin (Delphinus delphis) in the Mediterranean Sea and in the eastern North Atlantic Ocean. Conservation Genetics, 9: 1479-1487.

Öztürk, B. and Öztürk, A.A. (1997) Preliminary study on dolphin occurrence in the Turkish straits system. *European Research on Cetaceans* 11: 79-82.

Perrin, W.F. (2009) Common dolphins*: Delphinus delphis* and *D. capensis*. In *Encyclopedia of Marine Mammals*. Perrin, W.F., Wursig, B. and Thewissen, J.G.M. (eds). Academic Press, London. 2nd edition, pp. 255–259.

Politi, E. and Bearzi, G. (2001) Evidence of decline for a coastal common dolphin community in the eastern Ionian Sea. *European Research on Cetaceans* 15: 449-452.

Reid, J.B., Evans, P.G.H., and Northridge, S.P. (2003) Atlas of Cetacean Distribution in North-west European Waters. Joint Nature Conservation Committee, Peterborough. 76pp.

Ridoux, V., van Canneyt, O., Doremus, G., and Certain, G. (2003) Détermination des habitats préférentiels estivaux de prédateurs supérieurs pélagiques du proche Atlantique par observations aeriennes. Rapport contractuel CRMM-ULR/IFREMER, 19pp.

Rogan, E. and Mackey, M. (2007) Megafauna bycatch in drift-nets for albacore tuna (Thunnus alalunga) in the NE Atlantic. Fisheries Research, 86: 6–14.

Tregenza, N.J.C. and Collet, A. (1998) Common dolphin Delphinus delphis by-catch in pelagic trawl and other fisheries in the Northeast Atlantic. Reports of the International Whaling Commission, 48: 453-459.

Tregenza, N.J.C., Berrow, S.D., Leaper, R. and Hammond, P.S. (1997) Common Dolphin, Delphinus delphis L., by-catch in bottom set gillnets in the Celtic Sea. Reports of the International Whaling Commission, 47: 835-839.

Tudela, S., Kai Kai, A., Maynou, F., El Andalossi, F., and Guglielmi, P. (2004) Driftnet fishing and biodiversity: the case of the large-scale Moroccan driftnet fleet operating in the Alboran Sea (SW Mediterranean). *Biological Conservation,* 121: 65-78.

Vella A. (2002) Delphinus delphis (common dolphins) status in the central and southern Mediterranean around the Maltese islands. European Research on Cetaceans, 16: 124-127.

Zemsky, V.A. (1996) History of the Russian fishery of dolphins in the Black Sea. Pp. 46-48. In: B. Öztürk (ed.), Proceedings of the First International Symposium on the Marine Mammals of the Black Sea, pp. 120. (Istanbul, Turkey, 27-30 Jun 1994).

1.3 Harbour porpoise (*Phocoena phocoena*) in Europe

Step 1.1 - Biology of the species
The harbour porpoise (*Phocoena phocoena*) is the only member of the family Phocoenidae occurring in Europe. It is a small, relatively cryptic species that tends to avoid vessels, and rarely bow-rides. It usually occurs solitarily or in small groups of 2-3. Where food is concentrated or when making long-distance movements, larger temporary aggregations may form, numbering one hundred or more. Group composition appears to be fluid, except for adult females with dependent calves. Young are born mainly between May and August (peak in June), although some as early as March. The mating season is between April and September, with a peak in July-August. Porpoises reach sexual maturity at 3-5 years, with little difference between sexes. Once sexually mature, females may give birth to a single calf every 1-2 years. Longevity is relatively short, usually up to 12 years, although the maximum is 24 years in both sexes. Generation length has been estimated at 8.3 years (11.9 years in populations at a stable state). Diet varies both geographically and seasonally, and comprises small demersal, bentho- or epipelagic fish, such as whiting, sandeel, sprat, herring, cod, pouts and gobies. More detailed accounts of the biology of the species can be found in Bjørge & Donovan (1995), Donovan & Bjørge (1995), Read (1999), Evans *et al.* (2008), and Read & Tolley (2009).

Step 1.2 - Spatial scale of functioning
The harbour porpoise is found in temperate and sub-arctic seas of the northern hemisphere, occurring in both the Atlantic and Pacific. In the North Atlantic, the species occurs mainly from Central West Greenland and Novaya Zemlya in the north to North Carolina and Senegal in the south, with a geographically isolated population in the Black Sea (Reid *et al.* 2003, Evans *et al.* 2008, Fontaine *et al.* 2007, 2010). Although porpoises can be found in deep Atlantic waters off the edge of the continental shelf (for example within the Faroe Bank Channel), they are comparatively rare in waters exceeding 200 metres. The species frequently uses tidal conditions for foraging (see e.g., Evans 1997, Johnston *et al.* 2005, Pierpoint 2008, Marubini *et al.* 2009, Isojunno *et al.* 2012, Jones *et al.* 2014).

Figure 1.3.1 *North Atlantic Distribution of Harbour porpoise (depicting those areas where the species is thought to regularly occur)*

In European seas, it is common and widely distributed over the continental shelf (mainly at depths of 20-200m) from the Barents Sea and Iceland south to the coasts of France and Spain (Figure 1.3.1), although in the 1970s it became scarce in the southernmost North Sea, English Channel, and Bay of Biscay. Nevertheless, it remains the most widely distributed and frequently observed cetacean in North West European shelf seas, and since the 1990s, has returned to the southernmost North Sea, English Channel and French Biscay coast (Camphuysen 2004, Kiszka *et al.* 2004, 2007, Evans 2010, Hammond *et al.* 2013). It is largely absent from the Mediterranean (Frantzis *et al.* 2001), with small numbers reported mainly from the northern Aegean Sea, which probably come from the Black Sea population (Birkun & Frantzis 2008, Notarbartolo di Sciara & Birkun, 2010, Tonay & Dede 2013).

The isolated Black Sea harbour porpoise population is clearly genetically differentiated from Atlantic populations and has been classified as a separate subspecies, *Phocoena phocoena relicta* (Fontaine *et al.* 2007, 2010, 2017), but there is also genetic evidence that the Iberian population is distinct from porpoises further north; these are believed to have derived from an expansion of a small population off the northwest African coast (Fontaine *et al.* 2007, 2010). Porpoises along the French Atlantic coasts appear to be an admixture from the two genetically distinct populations along the Iberian coasts and in the North East Atlantic (Alfonsi *et al.* 2012). Further north, porpoises from southwestern UK appear to be differentiated from those in the North Sea (Fontaine *et al.* 2017).

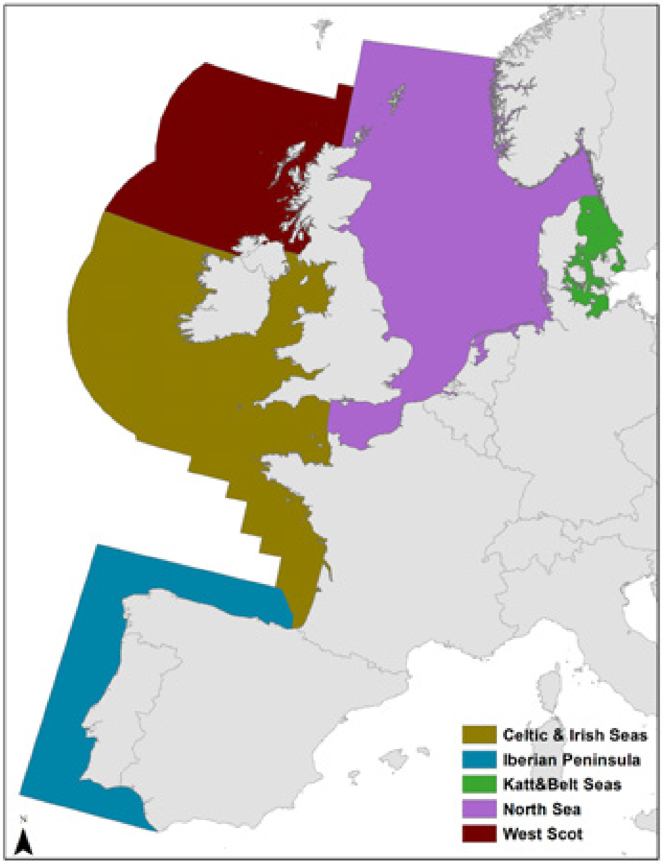


Figure 1.3.2 *Assessment Units for harbour porpoise in coastal Atlantic Europe (from ICES WGMME, 2013)*

Up to 16 separate Management Units (MU), thought to be demographically if not genetically distinct, have been proposed for the porpoise in the North Atlantic, using a precautionary approach of division rather than lumping where there is evidence to suggest this (Evans & Teilmann 2009, ICES 2012, Sveegaard *et al.* 2015, Fontaine *et al.* 2010, 2017, Lah *et al.* 2016). These include up to 11 MUs across North West Europe. These are: 1) the inner Baltic Sea (with its western boundary being around the Darss/Gedser underwater ridge or Rügen); 2) the southern Belt Sea; 3) the northern Belt Sea and southern Kattegat; 4) the northern Kattegat and Skagerrak; 5) the NE sector of the North Sea; 6) the western sector of the North Sea including the Eastern Channel; 7) the Celtic Sea along with South-west Ireland, the Irish Sea & Western Channel; 8) Northwest Ireland and Western Scotland; 9) the waters around the Faroe Islands south towards the Faroe-Shetland Channel; 10) the west French part of the Bay of Biscay; and 11) the Iberian Peninsula (coasts of Portugal and Atlantic Spain). The ICES Working Group on Marine Mammal Ecology (2014) has combined some of these into five assessment units (Figure 1.3.2). Further research is likely to refine and modify these subpopulations.

Various lines of evidence suggest that porpoises in the North Sea also exhibit some sub-structuring (Yurick & Gaskin 1987, Walton 1997, Lockyer 1999. Das *et al.* 2003, 2004, Teilmann *et al.*, 2008; Sveegaard *et al.*, 2011, De Luna *et al.* 2012). In particular, porpoises from the southwestern North Sea appear to differ significantly from those in the northeastern North Sea. There may be further sub-structuring between northern and southern North Sea and western vs eastern North Sea but it is difficult to determine where the division may lie given that different authors have used different sampling divisions and the precise origins of the samples are not necessarily known. Most studies indicate greater philopatry for females than male porpoises.

A number of authors allude to differences in ecology between animals from the northeastern and southern/western North Sea, particularly with respect to feeding. In recent years, seascape genetics has been used increasingly to account for differentiation observed between cetacean populations. In the absence of actual physical barriers, it has been shown in studies of various cetacean species that differences in ocean current systems and/or bathymetry parallel observed genetic differences. In the North Sea, the northeastern sector (from Shetland east to northern Denmark) is significantly deeper than the central and southernmost sectors. Current systems also show differences between the two regions both in terms of strength and circulation patterns. The central part of the North Sea represents an intermediate zone between the two regions, with respect to both bathymetry and ocean circulation.

If porpoises in the northeastern North Sea are feeding mainly upon pelagic prey (for which skull characteristics, particularly of the buccal cavity, have developed, as found by De Luna *et al.* 2012) whilst those in the southernmost North Sea are taking fish primarily off the bottom (with equivalent changes to the size of the buccal cavity), then these may be best considered separate management units with a potential boundary following bathymetric and oceanographic changes.

De Luna *et al.* (2012) and Andersen *et al.* (2001) also found significant differences between porpoises from the British North Sea and those from the Danish North Sea, as well as differences between porpoises from Norway and both the Danish North Sea and the British North Sea. And Wiemann *et al.* (2010) showed significant sub-structuring between the Danish North Sea and Norway. Thus, the presence of three Management Units within the North Sea may also be a possibility.

Within Danish waters, Wiemann *et al.* (2010) found clear genetic evidence of a population split between the Skagerrak and the Belt Seas, with a transition zone in the Kattegat area. This was particularly evident in significant frequency shifts of the most abundant mitochondrial haplotypes. A particular haplotype almost absent in the North Sea was the most abundant in the Belt Seas and Inner Baltic Sea. Microsatellites yielded a similar pattern.

Separation of the North Sea and the Baltic Sea populations with an east–west border within a transition zone in the Kattegat (waters South of the 58°N and North of 56°N latitudes) is supported by 3D skull geometric morphometric measurements (Galatius *et al.* 2012), satellite tracking (Sveegaard *et al.* 2011), as well as microsatellite and mitochondrial genetic analysis (Andersen *et al.* 2001, Wiemann *et al.* 2010). A combination of these studies suggests a transition zone between the Baltic Sea and Baltic Proper populations between Fehmarn Belt and 14°20’E (Sveegaard *et al.* 2015). The genetic evidence for a division between the Baltic Sea and the Baltic Proper populations is not as strong as the division between the North Sea and the Baltic Sea population, but Wiemann *et al.* (2010) nevertheless advocated a precautionary division into two separate Management Units. The less pronounced genetic and morphological differences may be due to the young age of the Baltic Proper population: a recent investigation indicates immigration and establishment of harbour porpoises in the Baltic Sea around 9000 years ago when the previously closed connection to the Atlantic Ocean was established through the Danish straits as a result of ice melting in the Arctic, following the last Ice Age (Sommer *et al.* 2008).

Most recently, Lah *et al.* (2016) undertook a genome-wide analysis, using single nucleotide polymorphisms (SNPs), derived from double digest restriction-site associated DNA sequencing (ddRAD-seq), as well as 13 microsatellite loci and mitochondrial haplotypes. They observed a distinct separation of the North Sea harbour porpoises from the Baltic Sea populations, but also identified splits between porpoise populations within the Baltic Sea. In particular, they observed a notable distinction between the Belt Sea and the Inner Baltic Sea sub-regions, supporting other authors who have advocated for porpoises in the Baltic Proper to be considered a distinct separate population.

From telemetry studies and passive acoustics, seasonal migrations of porpoises have been reported for the Skagerrak populations, which migrate westward in the winter towards the southern tip of Norway and into the northern North Sea, for the Kattegat populations which may migrate southward through the Great Belt in the winter, as well as for the porpoises from the Belt Sea and inner Baltic Sea, which may migrate westward to the Pomeranian Bay and from the Kadet trench into Danish waters, respectively (Verfuß *et al.* 2007, Sveegaard *et al.* 2011, Gallus *et al.* 2012, Benke *et al.* 2014).

Conclusions

Harbour porpoises are distributed more or less continuously across the shelf seas of northern Europe. Nevertheless, there is evidence (e.g. genetic, morphometric, telemetry studies of movements) for demographically distinct populations which we commonly refer to as management (or assessment) units. Animals around the Iberian Peninsula appear to be more closely related to those from NW Africa; in the Bay of Biscay the species is relatively uncommon and forms an admixture of animals from both south and north; the population in the Celtic Sea shows differences from those in the North Sea where there may be further sub-structuring, and separation from those in Danish Belt Seas. The porpoise population in the Baltic Proper is small, and distinct from those further west. The Black Sea population is geographically isolated from those in the Atlantic, the species being absent from much of the Mediterranean, and is genetically distinct. Presumably the species once occurred in the Mediterranean for there to have been colonisation of the Black Sea.

Harbour porpoise populations form at least ten management units in western Europe, and an isolated population in the Black Sea (with small numbers in the neighbouring Aegean Sea). Several of these occur across Member State boundaries, and therefore the species is perhaps best listed under category S1 as having a clearly sustainable meta-population above Member State level. Supra-national management units can be listed under category S4. It is not generally considered to be a fully migratory species, although there is movement of individuals across national boundaries.

Step 1.3 - Historical perspective: what happened to the species?

Although population surveys in Europe only started in the 1990s, several publications present evidence indicating widespread harbour porpoise declines during the middle of the last century (Verwey 1975, Teixeira 1979, Casinos & Vericad 1976, Evans 1980, 1990, 1992, Evans & Scanlan 1990, Smeenk 1987, Reijnders 1992, Tregenza 1992). Over that period, the species became rare in the southern North Sea, the English Channel, Bay of Biscay, and around the Iberian Peninsula. Since the 1990s, porpoises have returned in numbers to the southern North Sea and parts of the English Channel although they have become less abundant in the northernmost North Sea (Camphuysen 2004, Camphuysen & Peet 2006, Evans *et al.* 2003, 2008, Hammond *et al.* 2013).

There is uncertainty over the historical status of the species in the Mediterranean from which it is currently virtually absent, but during the latter half of the last century, numbers in the Black Sea were dramatically reduced by massive direct killing for the cetacean-processing industry that continued until 1983 (Smith 1982, IWC 2004, Birkun & Frantzis 2008). Large directed takes occurred during the years 1976-1983 before the ban on small cetacean hunting was declared in Turkey in 1983. Within that period, the total number of harbour porpoises killed was at least 163,000-211,000 (Birkun & Frantzis 2008). Illegal direct killing of unknown numbers continued in some parts of the Black Sea until 1991. Regionally extensive incidental mortality of porpoises in bottom-set gillnets is roughly estimated to have been in the thousands annually through the 1980s. The scale of this mortality almost certainly increased in the 1990s-2000s owing to the rapid expansion of illegal, unreported and unregulated fishing in the Black Sea (Birkun & Frantzis 2008).

Until 1983, unregulated hunting was the primary threat (IWC 1992, 2004). Very large numbers of harbour porpoises (as well as other cetaceans) were taken during the 20th century by all Black Sea countries for a variety of industrial uses (Kleinenberg 1956, Tomilin 1957). Although the total number killed is unknown, it may have been as many as four or five million for all species combined (e.g. see review in Smith 1982). It is widely accepted that all Black Sea cetacean populations, including Harbour Porpoises, were badly reduced by the directed fishery (IWC 1983, 1992, 2004). Catches of harbour porpoises were numerically fewer than those of common dolphins until 1964 when harbour porpoises became predominant (Danilevsky & Tyutyunnikov 1968, Smith 1982). Turkish catches of harbour porpoises in the early 1970s (see Berkes 1977) were thought to be at least as high as, and possibly much higher than, those estimated for 1976-1981 (34,000-44,000 per year according to IWC, 1983, assuming that harbour porpoises accounted for 80% of the total). At least since 1991, there has been no evidence of illegal directed takes although such takes had been reported before that time (IWC 1992).

At present, incidental mortality in bottom set gillnets is the most serious threat (*e.g.*, Birkun 2002a). Although all three Black Sea cetacean species are 'bycaught', the majority (95%) of recorded cetacean entanglements are of porpoises. Although absolute numbers of removals cannot be estimated from the available data., there are indications that the annual level of harbour porpoise bycatch may be in the thousands.

There have also been a number of mass mortality incidents of porpoises in the region. An explosion at a gas-drilling platform in the Azov Sea in August 1982 resulted in the deaths of over 2,000 porpoises (Birkun 2002b). Large-scale pelagic and small-scale coastal fisheries may affect Black Sea harbour porpoises indirectly by reducing their prey populations and degrading their habitat. Primarily, this relates to anchovies and sprats in the Black Sea and gobies in the Azov Sea. In particular, overfishing, eutrophication and the population explosion of an introduced predator, the ctenophore *Mnemiopsis leidyi*, led to a dramatic (8 to 12-fold) decline of sprat and anchovy abundance in the early 1990s (Prodanov *et al.* 1997). This reduced prey availability coincided with two mass mortality events (in 1989 and 1990) primarily affecting porpoises resulting from a severe nematode infection with bacterial complications (Birkun 2002c). This may have been exacerbated by immuno-suppression associated with PCB contamination Birkun & Frantzis 2008). Reported levels of DDTs and HCHs in Black Sea harbour porpoises at the time were higher than those in conspecifics elsewhere in the world (Tanabe *et al.* 1997). Chemical pollution is thus also a potential threat, particularly in the context of epizootics. Black Sea harbour porpoises are also affected in some years by ice entrapment in the Azov Sea

In the European Atlantic and North Sea, the main threat to harbour porpoise is thought to be by-catch, mainly from bottom set gill nets and pelagic trawls, with regions most affected including western Norway, central and southern North Sea, and Celtic Sea (Donovan & Bjørge 1995, Tregenza *et al.* 1997, Kaschner 2003, Vinther & Larsen 2004, Winship 2009, Bjørge *et al.* 2013, ICES 2015a, b). However, attacks by bottlenose dolphins (Ross & Wilson 1996, Jepson & Baker 1998), depletion of prey resources (Evans 1990, Reijnders 1992), noise disturbance (Carstensen *et al.* 2006, Brandt *et al.* 2011, Teilmann & Carstensen 2012), and infectious disease potentially arising from high contaminant levels (particularly of PCBs) (Bruhn *et al.* 1999, Kleivane *et* al 1999, Jepson *et al.* 2005, 2013, 2016, Hall *et al.* 2006) are also believed to pose threats. From post mortem examinations of 1,692 porpoises stranded in the UK between 1991 and 2010, 23% from infectious disease, 19% were thought to be the result of bottlenose dolphin attacks, 17% were diagnosed as by-catch, 15% died from starvation, and 4% live-stranded, (Deaville & Jepson 2011).

In inner Danish waters, harbour porpoises are thought to have been directly exploited since the Stone Age (Möhl 1970), with written records dating from the fourteenth century (Kinze 1995). The overall average annual take was thought to have been c. 1,000 animals with a known minimum total of 59,028 for the years 1819-92 (Kinze 1995). In the 1880s, the annual catch level increased to c. 2,000 of a grand total of c. 3,000 animals in all Danish waters. The actual catch may have been sustainable for several centuries until about 1870 (Kinze 1995). In the Lille Belt area, the take occurred annually in winter and spring, and consisted of migrating animals with summer ranges thought to extend into the Baltic proper. The population in the Baltic proper is currently estimated at only c. 600 animals (SAMBAH 2016), attributed to a marked decline in the second half of the twentieth century (Kinze 1995). The hunt ceased in the first half of the twentieth century although it is not clear whether the decline had already begun by then. The historic range of the harbour porpoise extended into the north-eastern parts of the Baltic Sea but during the second half of the twentieth century, when numbers declined, the distribution range narrowed, and now the species is largely confined to the western end (Koschinski 2002, SAMBAH 2016). It has often been concluded that porpoises escaped from ice cover in the eastern Baltic Sea in the winter and re-colonised the Baltic Proper in spring (Koschinski 2002). However, more recent observations, as described above, indicate that migration behaviour is much more complex and diffuse (Verfuß *et al.* 2007, Sveegaard *et al.* 2011, Gallus *et al.* 2012, Benke *et al.* 2014).

Concern has been raised about the long-term viability of the harbour porpoise population in the Baltic Proper. Bycatch at possibly unsustainable levels, contaminants, overfishing of prey species and disturbance have all been identified as potential threats (Koschinski 2002, SAMBAH 2016). Environmental contaminants most likely affect the long-term viability of Baltic Sea harbour porpoise stocks and might have been a major cause for the apparent decline between the 1940s and the 1970s (Teilmann & Lowry 1996, Koschinski 2002). Since then, concentrations of PCBs and other organochlorine contaminants have declined. To date, the most important threat to Baltic Sea harbour porpoises is bycatch in salmon drift nets and bottom-set gillnets (for cod and other demersal species) (Lindroth 1962, Skóra *et al.* 1988, Christensen 1991, Skóra 1991, Berggren 1994, Kock & Benke 1996), although noise pollution has the potential to increasingly become a threat due to the development of new activities in the Baltic Sea (e.g. offshore wind farms, fast ferries - Koschinski 2002, Teilmann & Carstensen 2012).

Climate change may also affect harbour porpoises either by altering human activities and thus pressures or by affecting the stock sizes and distribution of their prey (Evans & Bjørge, 2013). There is some evidence that changes in the stock sizes of initially herring and subsequently sand eel may have resulted in shifts in the distribution of porpoises in the North Sea and Danish waters (Evans 1990, Reijnders 1992, Hammond *et al.* 2013). In the past, porpoises in the Baltic Proper are reported to have experienced periodic catastrophic mortality from severe winter ice conditions (Lindroth 1962, Berggren 1994, Teilmann & Lowry 1996).

Step 1.4 - Analysis of distribution and trends

Population estimates do not exist for the entire North Atlantic range of the harbour porpoise, or even for the European range. However, the widest scale surveys were SCANS undertaken in 1994, followed by SCANS-II in 2005, and SCANS-III in 2016. From line transect SCANS surveys in July 1994 (Hammond *et al.* 2002), an overall population estimate of 341,000 porpoises (CV=0.14; 95% CI: 260,000-449,000) was made, with the following regional estimates: North Sea (c. 250,000), NW Scotland (c. 18,000), Baltic region (36,600 in Kattegat / Skagerrak / Belt Seas / Western Baltic Sea), Channel (0), and Celtic Shelf (36,300). Only a small portion of the Baltic Proper was surveyed.

A repeat survey in July 2005 (SCANS-II), covering a wider area (continental shelf seas from SW Norway, south to Atlantic Portugal), gave an estimate of 375,358 (CV=0.20; 95% CI: 256,300-549,700) (Hammond *et al.* 2013), with regional estimates: North Sea including Shetland & Orkney (c. 191,500), Baltic (19,100 in E. Skagerrak / Kattegat / Belt Seas / Western Baltic Sea), Channel (40.900), Celtic Shelf (72,400), Irish Sea (15,200), Atlantic Ireland & Scotland (33,800) and the Iberian Peninsula & SW France (2,400).

Comparing the two surveys, although the overall number estimated for the North Sea, Channel and Celtic Sea was comparable (341,000 in 1994, and 305,000 in 2005), numbers in the northern North Sea and Danish waters had declined from 239,000 to 120,000, whereas in the central and southern North Sea, Channel and Celtic Shelf, they had increased from 102,000 to 215,000. This is thought to represent a southward range shift rather than actual changes in population size (Winship 2009, Hammond *et al.* 2013), at least for the month of July. In Norwegian waters, estimates of 11,000 porpoises (95% CI: 4,790-25,200) for the Barents Sea and Norwegian waters north of 66^{O}N, and 82,600 (95% CI: 52,100-131,000) for southern Norway and the northern North Sea, were made during July 1989 (Bjørge & Øien 1995). There have been no abundance estimates for porpoises covering Norwegian waters since then.

A third SCANS (SCANS-III) survey was undertaken in July 2016. Taking equivalent areas between the three surveys, they revealed no significant change, with 407,177 (CV 0.18) in 1994, 519,864 (CV 0.21) in 2005/07, and 466,569 (CV 0.15) in 2016 (P.S. Hammond *pers. comm.*). The southwards shift in abundance of porpoises in the North Sea that was observed between 1994 and 2005 has persisted through to 2016.

The number of porpoises in the Baltic Proper has recently been estimated at 497 individuals (95% CI 80-1091), using sonar click detectors (C-PODs) deployed at a total of 304 locations spread around the region between 2011 and 2013 (SAMBAH 2016).

Conclusions

The harbour porpoise is thought to have been much more widespread and abundant in the shelf seas of Europe in previous centuries. Although quantitative information is largely lacking, the species appears to have experienced widespread declines in the middle of the twentieth century. Since the mid 1990s, populations of harbour porpoise in the Atlantic, North Sea and Danish Belt Seas may have remained stable. On the other hand, although direct hunting in those regions has ceased, the genetically distinct porpoise populations in the Baltic Proper and the Black Sea continue to face serious pressures from fisheries bycatch and a range of other human activities, with little signs of recovery. Current threats to populations throughout Europe include fisheries bycatch, depletion of prey resources, pollution and noise disturbance. If those human pressures are reduced, there is scope for recovery of depleted populations.

Ideally, population trends should be considered separately for each of the Management Units that have been recognised. Up to now, that has not been done although the three large-scale synoptic surveys conducted in northern Europe at decadal intervals have enabled abundance estimates to be made by ICES sub-area, and some of these more or less coincide with MUs. However, the confidence limits tend to be too wide at a regional scale to robustly determine trends. These surveys started in the mid-1990s after the Habitats Directive came into force. Thus, there are no data prior to human impacts in these areas, so it is not possible to set a historical baseline. Although the historical abundance and distribution is unknown, there is good evidence that the species was once more widely distributed in western Europe, and more abundant in the Baltic Proper and the Black Sea. The results of those surveys at a larger scale show no significant trend over the last twenty years.

Step 2.1 - FRP assessment

Scarcely any quantitative information exists of population numbers for bottlenose dolphin before the Habitats Directive came into force in 1992, despite the fact that this species is the best known of all cetaceans. Nevertheless, there is evidence that the species was more abundant in the Baltic Proper and Black Sea in historical times, and that in North-west Europe (particularly the North Sea and English Channel), it experienced widespread declines in the middle of the twentieth century from which it may then have partially recovered.

The indication therefore is that FRP for at least some of the management/assessment units should be above the current values. This is clearest for the Black Sea where habitat degradation and fisheries conflicts as well as past human exploitation have almost certainly depressed populations well below their natural carrying capacity. The same applies to the Baltic Proper which has been exposed to similar pressures. Thus, if conditions are improved, one might expect populations to increase, and to recolonise areas (e.g. eastern Mediterranean and eastern Baltic respectively), where the species is currently very rare. The recent shifts in centres of abundance observed in the North Sea very probably relate mainly to changes in food availability. Whether or not this affects the overall population size for different management/assessment units is unclear. Further south, there is a hiatus in the Bay of Biscay where the species is uncommon. This may relate primarily to the bathymetry of most of the Bay which is at greater depths to that of the favoured foraging habitat of porpoises. Recovery in this area may therefore be more constrained, although genetic studies indicate there is mixing of populations taking place along the French Atlantic coasts from both north and south.

As mentioned for bottlenose dolphin, whole genomic analysis using restriction site associated DNA sequencing is recommended for insights into the extent to which present day populations have experienced contractions in size and loss of genetic diversity. Genetic analysis enables one to estimate the effective population size (Ne) for management units prior to major human impacts.

The order of magnitude of FRP is difficult to determine but is likely to be in the range of one to five million animals. This is for the species throughout Europe and including the Black Sea. For individual MUs, it will be much less.

Conclusions

FRP >> CV (Baltic MUs in MBAL, Black Sea MUs in MBLS)
FRP > CV (Atlantic MUs in MATL)

Step 2.2 - FRR assessment

Harbour porpoises are thought to have occurred throughout the shelf seas of Europe. Nowadays it is largely absent from the Mediterranean Sea and the eastern part of the Baltic Proper. The current range could therefore extend to include those areas if suitable conditions prevailed.

Conclusions

FRR >> CV (Baltic MUs in MBAL, and Mediterranean MUs in MMED)
FRR > CV (Black Sea MUs in MBLS, Atlantic MUs in MATL)

References

Alfonsi, E., Hassani, S., Carpentier, F.-G., Le Clec'h, J.-Y., Dabin, W., Van Canneyt, O., Fontaine, M.C., and Jung, J.-K. (2012) A European melting pot of harbor porpoise in the French Atlantic coasts inferred from mitochondrial and nuclear data. PLoS One, 7(9): e44425. doi:10.1371/journal.pone.0044425.

Amundin, M., Wennerberg, D., Berglind, M., Hagberg, J, Konigson, S. and Lunneryd, S.- G. (2008) Undersökning av tumlarnärvaro i Skånes och Blekinges farvatten med hjälp av passiva akustiska tumlardetektorer och i samarbete med lokala yrkesfiskare (Study of the presence of harbour porpoises in southern Swedish coastal waters, as detected by passive acoustic loggers and in cooperation with local fishermen). Report to the Swedish Board of Fisheries.

Andersen, L.W., Ruzzante, D.E., Walton, M., Berggren, P., Bjørge, A. and Lockyer, C. (2001) Conservation genetics of the harbour porpoise, *Phocoena phocoena*, in eastern and central North Atlantic. *Conservation Genetics*, 2: 309-324.

Benke, H., Bräger, S., Dähne, M., Gallus, A., Hansen, S., Honnef, C.G., *et al.* (2014) Baltic Sea harbour porpoise pop- ulations: status and conservation needs derived from recent survey results. *Marine Ecology Progress Series*, 495: 275–290. doi: 10.3354/meps10538.

Berrow, S.D., Long, S.C., McGarry, A.T., Pollard, D., Rogan, E., and Lockyer, C. (1998) Radio-nuclides (Cs-137 and K-40) in harbour porpoises *Phocoena phocoena* from British and Irish coastal waters. *Marine Pollution Bulletin*, 36: 569-576.

Berkes, F. 1977. Turkish dolphin fisheries. *Oryx* 14(2): 165-167.

Birkun, Jr. A. 2002a. Interaction between cetaceans and fisheries: Black Sea. In: G. Notarbartolo di Sciara (ed.), *Cetaceans of the Mediterranean and Black Seas: State of Knowledge and Conservation Strategies*, pp. 98-107. ACCOBAMS Secretariat, Monaco.

Birkun Jr., A. 2002b. Disturbance: Black Sea. Pp. 161-166. In: G. Notarbartolo di Sciara (ed.), *Cetaceans of the Mediterranean and Black Seas: State of Knowledge and Conservation Strategies*. ACCOBAMS Secretariat, Monaco.

Birkun Jr., A. 2002c. Natural mortality: Black Sea. In: G. Notarbartolo di Sciara (ed.), *Cetaceans of the Mediterranean and Black Seas: State of Knowledge and Conservation Strategies*, pp. 181-193. ACCOBAMS Secretariat, Monaco.

Birkun, A.A. and Frantzis, A. (2008) *Phocoena phocoena ssp. Relicta.* In IUCN (2008). IUCN Red List of Threatened Species. www.iucnredlist.org.

Bjørge, A. and Donovan, G.P. (editors) (1995) Biology of the Phocoenids. *Reports of the International Whaling Commission* (Special Issue No. 16: 1-552.

Bjørge, A. and Øien, N. (1995) Distribution and abundance of harbour porpoise, *Phocoena phocoena*, in Norwegian waters. *Reports of the International Whaling Commission* (Special Issue No. 16): 89-98.

Brandt, M.J., Diederichs, A., Betke, K., and Nehls, G. (2011) Responses of harbour porpoise to pile driving at the Horns Rev II offshore wind farm in the Danish North Sea. *Marine Ecology Progress Series*, 421: 205–216.

Bruhn, R., Kannan, N., Petrick, G., Schulz-Bull, D.E., and Duinker, J.C. (1999) Persistent chlorinated organic contaminants in harbour porpoises from the North Sea and Arctic waters. *The Science of the Total Environment*, 237-238: 351-361.

Camphuysen, C.J. (2004) The return of the harbour porpoise (*Phocoena phocoena*) in Dutch coastal waters. *Lutra*, 47: 113-122.

Camphuysen C.J. and G. Peet (2006) *Whales and dolphins in the North Sea*. Fontaine Uitgevers, Kortenhoef, The Netherlands.

Carstensen J., Henriksen O.D., and Teilmann J. (2006) Impacts of offshore wind farm construction on harbour porpoises: acoustic monitoring of echolocation activity using porpoise detectors (T-PODs). *Marine Ecology Progress Series*, 321: 295–308.

Casinos, A. and Vericad, J.-R. (1976) The cetaceans of the Spanish coasts: a survey. *Mammalia,* 40: 267-289.

Christensen, O. (1991) Bycatches in the salmon drift net fishery in the Baltic Sea. *International Council for the Exploration of the Sea, Anadromous and Catadromous Fish Committee, C.M.* 1991/M: 26, Ref. J+N. 9 pp. typescript.

Danilevsky, N.N. and Tyutyunnikov, V.P. (1968) Some data on the current state of dolphin stocks in the Black Sea. *Rybnoye Khozyaystvo* N11: 25-27 (in Russian).

Das, K. Lepoint, G., Leroy, Y., and Bouquegneau, J.M. (2003) Marine mammals from the southern North Sea: feeding ecology data from δ^{13}C and δ^{15}N measurements. *Marine Ecology Progress Series*, 263: 287-298.

Das, K., Siebert, U., Fontaine, M., Jauniaux, T., Holsbeek, L., and Bouquegneau, J.M. (2003) Ecological and pathological factors related to trace metal concentrations in harbour porpoises *Phocoena phocoena* from the North Sea and adjacent waters. *Marine Ecology Progress Series*, 281: 283-295.

De Luna L., C.J. (2005) *The relationship between developmental stability, genomic diversity and environmental stress in two cetacean species: The harbour porpoise (Phocoena phocoena) and the bottlenose dolphin (Tursiops truncatus*). Ph.D Thesis, Univesity of Durham, UK, 204pp.

De Luna, C.J., Goodman, S.J., Thatcher, O., Jepson, P.D., Tolley, K., and Hoelzel, A.R. (2012) Phenotypic and genetic divergence among harbour porpoise populations associated with habitat regions in the North Sea and adjacent seas. *Journal of Evolutionary Biology*, 25: 674-681.

Donovan, G. P. and Bjørge, A. (1995) Harbour porpoises in the North Atlantic: edited extract from the Report of the IWC Scientific Committee, Dublin 1995. *Reports of the International Whaling Commission* 16: 3-25.

Evans, P.G.H. (1980) Cetaceans in British Waters. *Mammal Review*, 10: 1-52.

Evans, P.G.H. (1990) European cetaceans and seabirds in an oceanographic context. *Lutra*, 33: 95-125.

Evans, P.G.H. (1992) *Status Review of Cetaceans in British and Irish Waters.* UK Dept. of the Environment, London. 98pp.

Evans, P.G.H. and Bjørge, A. (2013) Impacts of climate change on marine mammals. Marine Climate Change Impacts Partnership (MCCIP) Science Review 2013, 134-148. Published online 28 November 2013 doi:10.14465/2013.arc15.134-148.

Evans, P.G.H. and Scanlan, G. (1990) Historical Status Changes of Cetaceans in British and Irish waters. *European Research on Cetaceans*, 3: 51-57.

Evans, P.G.H. and Teilmann, J. (Editors) (2009) *Report of ASCOBANS/HELCOM Small Cetacean Population Structure Workshop*. ASCOBANS/UNEP Secretariat, Bonn, Germany. 140pp.

Evans, P.G.H., Anderwald, P. and Baines, M.E. (2003) *UK Cetacean Status Review.* Report to English Nature and Countryside Council for Wales. Sea Watch Foundation, Oxford. 160pp.

Evans, P.G.H., Lockyer, C.H., Smeenk, C., Addink, M., and Read, A.J. (2008) Harbour Porpoise *Phocoena phocoena*. Pp.704-709. In: *Mammals of the British Isles*. (Eds. S. Harris and D.W. Yalden). Handbook. 4th Edition. The Mammal Society, Southampton. 800pp.

Fontaine, M.C., Baird, S.J., Piry, S., Ray, N., Tolley, K.A., Duke, S., Burkun, A., Jr., Ferreira, M., Jauniaux, T., Llavona, A., Ozturk, B., Ozturk, A.A., Ridoux, V., Rogan, E., Sequeira, M., Siebert, U., Vikingsson, G.A., Bouquegneau, J.M. and Michaux, J.R. (2007) Rise of oceanographic barriers in continuous populations of a cetacean: the genetic structure of harbour porpoises in Old World waters. *BMC Biology*, 5: 1-16.

Fontaine, M.C., Thatcher, O., Ray, N., Piry, S., Brownlow, A., Davison, N.J., Jepson, P., Deaville, R., and Goodman, S.J. (2017) Mixing of porpoise ecotypes in southwestern UK waters revealed by genetic profiling. *Royal Society Open Science*, 4: 160992. http://dx.doi.org/10.1098/rsos.160992.

Fontaine, M.C., Tolley, K.A., Michaux, J.R., Birkun Jr., A., Ferreira, M., Jauniaux, T., Llavona, Ń., Oztürk, B., Oztürk, A.A., Ridoux, V., Rogan, E., Sequeira, M., Bouquegneau, J.-M. and Baird, S.J.E. (2010) Genetic and historic evidence for climate-driven population fragmentation in a top cetacean predator: The harbour porpoises in European waters. *Proceedings of the Royal Society B: Biological Sciences*, 277 (1695): 2829-2837.

Frantzis, A., Gordon, J., Hassidis, G., and Komnenou, A. (2001) The enigma of harbor porpoise presence in the Mediterranean Sea. *Marine Mammal Science*, 17(4), 937-944.

Galatius, A., Kinze, C.C., and Teilmann, J. (2012) Population structure of harbour porpoises in the Baltic region: evidence of separation based on geometric morphometric comparisons. *Journal of the Marine Biological Association U K*., 92: 1669–1676. doi: 10.1017/s0025315412000513.

Gallus, A., Dähne, M., Verfuß, U.K., Bräger, S., Adler, S., Siebert, U., *et al.* (2012) Use of static passive acoustic monitoring to assess the status of the “Critically Endangered” Baltic harbour porpoise in German waters. *Endangered Species Research*, 18: 265–278. doi: 10.3354/esr00448.

Hall, A.J., Hugunin, K., Deaville, R., Law, R.J., Allchin, C.R., and Jepson, P.D. (2006) The risk of infection from polychlorinated biphenyl exposure in harbour porpoise (*Phocoena phocoena*) – a case-control approach. *Environmental Health Perspectives*, 114: 704-711.

Hammond, P.S., Berggren, P., Benke, H., Borchers, D.L., Collet, A., Heide-Jørgensen, M.P., Heimlich, S., Hiby, A.R., Leopold, M.F. and Øien, N. (2002) Abundance of harbour porpoise and other cetaceans in the North Sea and adjacent waters. *Journal of Applied Ecology*, 39: 361-376.

Hammond, P.S., Macleod, K., Berggren, P., Borchers, D.L., Burt, M.L., Cañadas, A., Desportes, G., Donovan, G.P., Gilles, A., Gillespie, D., Gordon, J., Hiby, L., Kuklik, I., Leaper, R., Lehnert, K., Leopold, M., Lovell, P., Øien, N., Paxton, C.G.M., Ridoux, V., Rogan, E., Samarra, F., Scheidat, M., Sequeira, M., Siebert, U., Skov, H., Swift, R., Tasker, M.L., Teilmann, J., Van Canneyt, O. and Vázquez, J.A. (2013) Cetacean abundance and distribution in European Atlantic shelf waters to inform conservation and management. *Biological Conservation,* 164**:** 107–122.

ICES (2013) Report of the Working Group on Marine Mammal Ecology (WGMME), 4–7 February 2013, Paris, France. ICES CM 2013/ACOM:26. 117pp.

ICES (2015a) Bycatch of small cetaceans and other marine animals – Review of national reports under Council Regulation (EC) 812/2004 and other published documents. In: Report of the ICES Advisory Committee, 2015. ICES Advice 2015, Book 1, Section 1.6.1.1.

ICES (2015b) Report of the Working Group on Bycatch of Protected Species (WGBYC), 2-6 Feb 2015, ICES Headquarters, Copenhagen, Denmark. ICES CM 2015\ACOM:26. 82pp.

Isojunno, S., Matthiopoulos, J., and Evans, P.G.H. (2012) Harbour porpoise habitat preferences: Robust spatio-temporal inferences from opportunistic data. *Marine Ecology Progress Series*, 448: 155-170.

IWC (International Whaling Commission) (1996) Report of the Sub-Committee on Small Cetaceans. *Reports of the International Whaling Commission*, 46: 160-179.

IWC (1983) Report of the Sub-Committee on Small Cetaceans. *Report Of The International Whaling Commission* 33: 152-170.

IWC (1992) Report of the Sub-Committee on Small Cetaceans. *Report Of The International Whaling Commission* 42: 178-234.

IWC (2004) Annex L. Report of the Sub-committee on Small Cetaceans. *Journal of Cetacean Research and Management* 6(Suppl.): 315-334.

Jepson, P.D. and Baker, J.R. (1998) Bottlenosed dolphins (*Tursiops truncatus*) as a possible cause of acute traumatic injuries in porpoises (*Phocoena phocoena*). *Veterinary Record*, 143: 614-615.

Jepson, P.D., Deaville, R., Barber, J.L., Aguilar, A., Borrell, A., Murphy, S., Barry, J., Brownlow, A., Barnett, J., Berrow, S., Cunningham, A.A., Davison, N.J., ten Doeschate, M., Esteban, R., Ferreira, M., Foote, A.D., Genov, T., Giménez, J., Loveridge, J., Llavona, A., Martin, V., Maxwell, D.L., Papachlimitzou, A., Penrose, R., Perkins, M.W., Smith, B., de Stephanis, R., Tregenza, N., Verborgh, P., Fernandez, A., and Law, R.J. (2016). PCB pollution continues to impact populations of orcas and other dolphins in European waters. *Nature Scientific Reports*. | 6:18573 | DOI: 10.1038/srep18573.

Jepson, P.D., Deaville, R., Barnett, J., Davison, N.I., Paterson, I.A.P., Penrose, R.S., Perkins, M.P., Reid, R.J., and Law, R.J. (2013). Persistent organic pollutants in UK stranded cetaceans (1990-2008): linking exposure with toxic effects. In: P.G.H. Evans (ed.) *Chemical Pollution and Marine Mammals*. European Cetacean Society Special Publication 55: 18-24.

Johnston, D.W., Westgate, A.J., and Read, A.J. (2005) Effects of fine-scale oceanographic features on the distribution and movements of harbour porpoises *Phocoena phocoena* in the Bay of Fundy. *Marine Ecology Progress Series*, 295: 279–293. http://www.int-res.com/abstracts/meps/v295/p279-293/.

Jones, A.R., Hosegood, P., Wynn, R.B., De Boer, M.N., Butler-Cowdry, S., and Embling, C. B. (2014) Fine-scale hydrodynamics influence the spatio-temporal distribution of harbour porpoises at a coastal hotspot. *Progress in Oceanography*, 128: 30–48. http://www.sciencedirect.com/science/article/pii/S0079661114001256.

Kaschner, K. (2003) *Review of small cetacean by-catch in the ASCOBANS area and adjacent waters – current status and suggested future actions*. ASCOBANS Report MOP4/Doc22(s) presented at the 4th Meeting of the Parties to ASCOBANS, Esbjerg, Denmark. 122pp.

Kinze, C.C. (1995) Exploitation of harbour porpoises (*Phocoena phocoena*) in Danish waters: a historical review. *Reports of the International Whaling Commission* (Special Issue 16): 141-153.

Kleinenberg, S.E. (1956) *Mammals of the Black and Azov Seas: Research Experience for Biology and Hunting*. USSR Academy of Science Publishing House, Moscow. (in Russian).

Kiszka, J.J., Hassani, S., and Pezeril, S. (2004) Distribution and status of small cetaceans along the French Channel coasts: using opportunistic records for a preliminary assessment. *Lutra*, 47: 33–46.

Kiszka, J., Macleod, K., Van Canneyt, O, Walker, D., and Ridoux, V. (2007) Distribution, encounter rates, and habitat characteristics of toothed cetaceans in the Bay of Biscay and adjacent waters from platform-of-opportunity data. *ICES Journal of Marine Science*, 64: 1033-1043.

Kleivane, L., Skaare, J.U., Bjørge, A., de Ruiter, E., and Reijnders, P.J.H. (1995) Organochlorine pesticide residue and PCBs in harbour porpoise (*Phocoena phocoena*) incidentally caught in Scandinavian waters. *Environmental Pollution*, 89: 137-146.

Kock, K.-H., and Benke, H. (1996) On the bycatch of harbour porpoise (*Phocoena phocoena*) in German fisheries in the Baltic and the North Sea. *Archive of Fishery and Marine Research,* 44(1/2): 95-114.

Koschinski, S (2002) Current knowledge of the harbour porpoises (*Phocoena phocoena*) in the Baltic Sea. *Ophelia*, 55: 167-197.

Lah, L., Trense, D. Benke, H., Berggren, P**.,** Gunnlaugsson, Þ**.,** Lockyer, C., Özturk, A., Özturk, B., Pawliczka, I., Roos, A., Siebert, U., Skóra, K., Vikingsson, G., and Tiedemann, R. (2016) Spatially explicit analysis of genome-wide SNPs detects subtle population structure in a mobile marine mammal, the harbor porpoise. PLoS ONE 11(10): e0162792. doi:10.1371/journal.pone.0162792.

Lindroth, A. (1962) Baltic salmon fluctuations 2: porpoise and salmon. *Report of the Institute for Freshwater Research Drottningholm* 44: 105-112.

Lockyer, C. (1999) Application of a new method to investigate population structure in the harbour porpoise, *Phocoena phocoena*, with special reference to the North and Baltic Seas. *Journal of Cetacean Research and Management*, 1(3): 297-304.

Marubini, F., Gimona, A., Evans, P.G.H., Wright, P.J. and Pierce, G.J. (2009) Habitat preferences and interannual variability in occurrence of the harbour porpoise *Phocoena phocoena* in the north-west of Scotland. *Marine Ecology Progress Series,* 381: 297-310.

Pierpoint C (2008) Harbour porpoise (*Phocoena phocoena*) foraging strategy at a high energy, near-shore site in south-west Wales, UK. *Journal of the Marine Biological Association UK*, 88: 1167–1173.

Prodanov, K., Mikhailov, K., Daskalov, G., Maxim, C., Chashchin, A., Arkhipov, A., Shlyakhov, V., Ozdamar, E. 1997. Environmental Management of Fish Resources in the Black Sea and their Rational Exploitation. Studies and Reviews of the General Fisheries Council for the Mediterranean. FAO, Rome.

Read, A.J. (1999) Harbour porpoise *Phocoena phocoena* (Linneaus, 1758). In: S. H. Ridgway and R. Harrison (eds), Handbook of marine mammals, Vol. 6: The second book of dolphins and the porpoises. Academic Press., London, pp. 323-356.

Read, A.J. and Tolley, K. (2009) Harbor Porpoise *Phocoena phocoena*. In: *Encyclopedia of Marine Mammals*. Perrin, W.F., Wursig, B. and Thewissen, J.G.M. (eds). Academic Press, London. 2nd edition, pp. 530–533.

Reid, J.B., Evans, P.G.H., & Northridge, S.P., (2003) *Atlas of Cetacean distribution in north-west European waters*. 76pp. ISBN 1 86107 550 2.

Reijnders, P.J.H. (1992) Harbour porpoises, *Phocoena phocoena*, in the North Sea: numerical responses to changes in environmental conditions. *Netherlands Journal of Aquatic Ecology*, 26, 75-85.

Ross, H.M. and Wilson, B. (1996) Violent interactions between bottlenose dolphins and harbour porpoises. *Proceedings of the Royal Society London*, 263: 283–286.

Skóra, K.E. (1991) Notes on cetacea observed in the Polish Baltic Sea: 1979-1990. *Aquatic Mammals,* 17: 67-70.

Skóra, K.E., Pawliczka, I. and Klinowska, M. (1988) Observations of the harbour porpoise (*Phocoena phocoena*) on the Polish Baltic coast. *Aquatic Mammals* 14(3): 113- 119.

Smeenk, C. (1987) The harbour porpoise *Phocoena phocoena* (L., 1758) in the Netherlands: stranding records and decline. *Lutra*, 30: 77-90.

Smith, T.D. (1982) Current understanding of the status of the porpoise populations in the Black Sea. *Mammals in the Seas, Vol. 4: FAO Fisheries Series* 5(4): 121-130.

Sommer, R.S., Pasold, J., and Schmolcke, U. (2008) Post-Glacial immigration of the harbour porpoise (*Phocoena phocoena*) into the Baltic Sea. *Boreas*, 37: 458–464.

Sveegaard, S., Teilmann, J., Tougaard, J., Dietz, R., Mouritsen, K.N., Desportes, G. and Siebert, U. (2011) High density areas for harbour porpoises (*Phocoena phocoena*) identified by satellite tracking. *Marine Mammal Science*, 27: 230-246.

Sveegaard, S., Galatius, A., Dietz, R., Kyhn, L., Koblitz, J.C., Amundin, M., Nabe-Nielsen, J., Sindling, M.-H.S., Andersen, L.W., and Teilmann, J. (2015) Defining management units for cetaceans by combining genetics, morphology, acoustics and satellite tracking. *Global Ecology and Conservation*, 3: 839–850. doi: 10.1016/j.gecco.2015.04.002.

Tanabe, S., Madhusree, B., Öztürk, A.A., Tatsukawa, R., Miyazaki, N., Özdamar, E., Aral, O., Samsun, O. and Öztürk, B. (1997) Persistent organochlorine residues in harbour porpoise (*Phocoena phocoena*) from the Black Sea. *Marine Pollution Bulletin,* 34(5): 338-347.

Teilmann, J. and Carstensen, J. (2012) Negative long term effects on harbour porpoises from a large scale offshore wind farm in the Baltic - evidence of slow recovery. *Environmental Research Letters*, 7: doi:10.1088/1748-9326/7/4/045101.

Teilmann, J., and Lowry, N. (1996) Status of the harbour porpoise (*Phocoena phocoena*) in Danish waters. *Report of the International Whaling Commission,* 46: 619-625.

Teixeira, A.M.A.P. (1979) Marine mammals of the Portuguese coast. *Z. Säugetierkunde*, 44: 221-238.

Tomilin, A.G. (1957) *Mammals of the USSR and Adjacent Countries. Vol. IV. Cetaceans*. USSR Academy of Science Publishing House, Moscow. (in Russian).

Tonay, A.M. and Dede, A. (2013) First stranding record of a harbor porpoise (Phocoena phocoena) in the southern Aegean Sea. J. Black Sea / Mediterr. Environ. 19, 132-137.

Tregenza, N.J.C. (1992) Fifty years of cetacean sightings from the Cornish coast, SW England. *Biological Conservation,* 59: 65-70.

Tregenza, N.J.C., Berrow, S.D., Hammond, P.S. and Leaper, R. (1997) Harbour porpoise (*Phocoena phocoena* L.) by-catch in set gillnets in the Celtic Sea. *ICES Journal of Marine Science*, 54: 896-904.

Verfuß, U.K., Honnef, C.G., Meding, A., Dähne, M., Mundry, R., and Benke, H. (2007) Geographical and seasonal variation of harbour porpoise (*Phocoena phocoena*) presence in the German Baltic Sea revealed by passive acoustic monitoring. *Journal of the Marine Biological Association UK*, 87: 165-176. doi: 10.1017/s0025315407054938.

Verwey, J. (1975) The cetaceans *Phocoena phocoena* and *Tursiops truncatus* in the Marsdiep area (Dutch Waddensea) in the years 1931-1973. *Nederlands Instituut voor Onderzoek der Zee,* 17a, b: 1-153.

Vinther, M. and Larsen, F. (2004) Updated estimates of harbour porpoise by-catch in the Danish bottom set gillnet fishery. *Journal of Cetacean Research and Management*, 6(1): 19-24.

Walton, M.J. (1997) Population structure of harbour porpoises *Phocoena phocoena* in the seas around the UK and adjacent waters. *Proceedings of the Royal Society of London* Series B, 264: 89-94.

Wiemann, A., Andersen, L.W., Berggren, P., Siebert, U., Benke, H., Teilmann, J., Lockyer, C., Pawliczka, I., Skóra, K., Roos, A., Lyrholm, T., Paulus, K.B., Ketmaier, V., and Tiedemann, R. (2010) Mitochondrial Control Region and microsatellite analyses on harbour porpoise (*Phocoena phocoena*) unravel population differentiation in the Baltic Sea and adjacent waters. *Conservation Genetics*, 11: 195-211.

Winship, A.J. (2009) *Estimating the impact of bycatch and calculating bycatch limits to achieve conservation objectives as applied to harbour porpoise in the North Sea*. PhD thesis. University of St Andrews, UK.

Yurick, D.B. and Gaskin, D.E. (1987) Morphometric and meristic comparisons of skulls of harbour porpoise *Phocoena phocoena* (L.) from the North Atlantic and the North Pacific. *Ophelia*, 27: 53-75.

1.4 Humpback whale (*Megaptera novaeangliae*) in the European Atlantic

Step 1.1 - Biology of the species

The humpback whale (*Megaptera novaeangliae*) is a large rorqual from the family Balaenopteridae that includes blue, fin, sei and minke whale. The species has no strongly organised social structure. It is typically found singly or in small unstable groups, although large feeding aggregations can occur in summer, and large competitive groups of males can form around females in breeding areas. Group size is often associated with the size of the exploited prey patch, although due to the severe historical overexploitation of the species, aggregations rarely number more than ten individuals. Breeding is strongly seasonal, with most births in low latitudes in winter (peak January-March). The age at which humpbacks become sexually mature varies by population, from 4 to >10 years for both sexes. Once mature, females give birth to a single calf typically every 2-3 years, with occasional annual calving. Longevity is at least 48 years, and possibly much longer. The generation length has been estimated at 14.5 years (21.5 years in populations at a stable state). The generalist diet includes both shoaling fish, notably herring, sprat, sandeel, mackerel and, in polar regions, capelin, and plankton (mainly euphausiids). More detailed accounts of the biology of the species can be found in Clapham (1996), Claphan & Mead (1999), Clapham & Evans (2008), and Clapham (2009).

Step 1.2 - Spatial scale of functioning

The humpback whale has a worldwide distribution in all seas, occurring even occasionally to the ice edge. It is a highly migratory species, feeding in summer in high latitudes, and mating and calving in winter in tropical waters, although a few overwinter on the feeding grounds. The species shows strong individual fidelity to feeding areas; in the North Atlantic, these include the Gulf of Maine, Gulf of St Lawrence, Newfoundland/Labrador, Greenland, Iceland and Norway. Matching of photographically and genetically identified individuals indicates that the eastern North Atlantic population migrates primarily to the West Indies (Martin *et al.* 1984, Stevick *et al.* 1998, 2003, 2006), although some animals winter in the Cape Verde Islands (Reiner *et al.* 1996, Hazevout & Wenzel 2000, Jann *et al.* 2003); genetic analysis suggests a third, unknown, breeding area. Despite fidelity to specific feeding grounds, however, whales from all North Atlantic areas appear to mix spatially and genetically in the West Indies in winter.

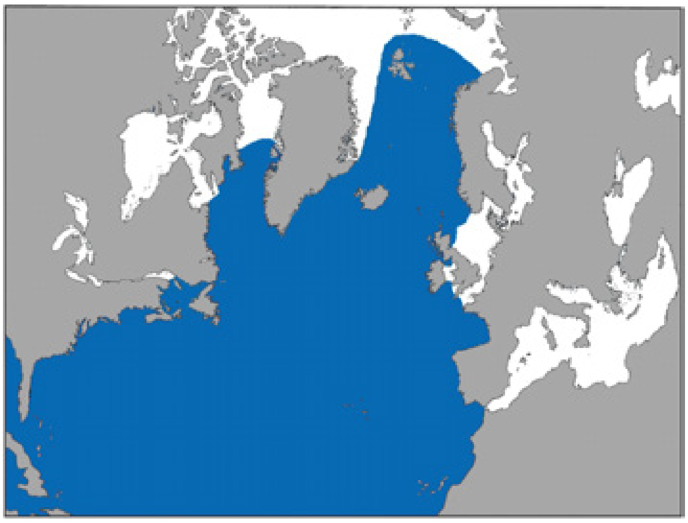


Figure 1.4.1 *North Atlantic distribution of humpback whale (depicting those areas where the species is thought to regularly occur)*

The species is more common on the west side, spending the summer in Baffin Bay and along the New England coast. In the eastern North Atlantic it occurs around Iceland, Norway, the British Isles and Ireland. North Atlantic humpbacks winter in the West Indies with small numbers around the Cape

Verde Islands off North-west Africa (Figure 1.4.1). The species is only a casual visitor to the Mediterranean and the Baltic Seas (Notarbartolo di Sciara & Birkun 2010, Kinze *et al.* 2010).

In European shelf waters, humpback whales occur mainly between May and September, and this was the period when catches from whaling activities were highest (Thompson 1928, Brown 1976, Clapham & Evans 2008). However, particularly from the British Isles and Ireland south to the Iberian Peninsula, sightings may occur at any time of year, including November to March, as indicated also from acoustic detections with SOSUS hydrophone arrays offshore in the North Atlantic at these latitudes (Clark & Charif 1998, Charif *et al.* 2001). Sightings in Ireland, occurring mainly along the south coast, increase through the summer to peak in September to December, rapidly declining between January and May (Berrow *et al.* 2010).

Six distinct feeding aggregations have been identified in the North Atlantic: Gulf of Maine; Gulf of St Lawrence; Newfoundland/Labrador; West Greenland; Iceland; and North Norway (including Bear Island and Jan Mayen) (Reilly *et al.* 2008). Genetic and photo-ID data indicate that the six feeding aggregations represent relatively discrete subpopulations, fidelity to which is determined matrilineally. However, because whales from different feeding grounds all mix in a common breeding area in the West Indies, there is male-mediated nuclear gene flow between the subpopulations.

Conclusions

Current knowledge indicates that the humpback whale in the Atlantic, which is its main range in Europe, should be listed under category MNR4, a non-reproductive population of a migratory species with individuals with large home ranges and showing large cyclic, directed movements, although it is possible that a segment of the North Atlantic population belongs in category MNR1, as a largely non-reproductive population of a (partial) migratory species with clearly sustainable size and extent above Member State. The same population can occur in different Member States in time.

Step 1.3 - Historical perspective: what happened to the species?

Humpbacks were heavily exploited in the past by pre-modern whaling in their breeding grounds in both the West Indies and the Cape Verde islands, and by modern whaling in their feeding grounds, especially off Iceland and off Norway in the late 19th and early 20th centuries (Reilly *et al.* 2008). Catches of pre-modern whaling are estimated primarily from trade records. Catches of early modern whaling also need to be estimated, because most of the catch records were not divided by species. Catches in the West Indies (including Bermuda) are documented from 1664 to the present day, but the main period was 1826-1928, during which about 8,600 whales were estimated to have been killed. Whaling in the Cape Verde Islands occurred primarily during 1850-1912 with a total estimated kill of about 3,000 animals. An estimated 3,200 were taken from Iceland and 2,000 from northern Norway during 1880-1916 (Reilly *et al.* 2008). About 1,500 humpback whales are reported to have been killed in the North Atlantic since 1916, from a variety of areas including the British Isles, Faroe Islands, Norway, Iceland, Greenland, and eastern Canada, as well as Norwegian pelagic catches (Reilly *et al.* 2007). However, humpbacks are no longer exploited in Europe.

There remains much debate over what the pre-exploitation population sizes were for humpback whale (and other baleen whales), due to inaccuracies in the catch record, uncertainties surrounding genetic estimates, and/or differences in time scales applied to the estimates (Roman & Palumbi 2003, Holt & Mitchell 2004, Punt *et al.* 2006, Alter & Palumbi 2009, Smith & Reeves 2010, Ruegg *et al.* 2013). Nevertheless, they generally indicate marked reductions in population sizes for most of the baleen whale species since commercial whaling started. Based upon mtDNA analyses of genetic diversity, for example, the pre-exploitation population size of North Atlantic humpback whales has been estimated at 240,000 compared with a "current" estimate of 9,300-12,100 (Roman & Palumbi 2003). Ruegg *et al.* (2013), estimated the long-term population size of c. 112,000 individuals, based on nuclear gene diversity, which although lower than the mtDNA estimate, is still 2-3x higher than estimates based upon catch data. Commercial whaling of humpback whales in the North Atlantic was banned by the IWC in 1955, by which time it was estimated that this population had been reduced to 1,000 individuals (Mitchell & Reeves 1983, Katona & Beard 1990). So, whatever the historical population size, humpbacks in the North Atlantic do appear to be steadily increasing though probably still nowhere near pre-exploitation levels.

The other known causes of mortality are bycatch, mainly through entanglement in ghost netting or creel lines (Northridge *et al.* 2010; Ryan *et al.*, 2016) and ship strikes (Evans *et al.* 2011), although the species may be affected by noise disturbance from seismic surveys (Risch *et al.* 2012, Cerchio *et al.*, 2014). Prey depletion through overfishing or the effects of climate change may also have an influence on humpback whale abundance and distribution (Evans & Bjørge 2013).

Step 1.4 - Analysis of distribution and trends

Overall, the North Atlantic population has recovered well from exploitation, estimated at somewhere between 9,400-16,400 in 1992, with the great majority occurring in the west-central part (Smith *et al.* 1999, Stevick *et al.* 2003). NASS surveys around Iceland in the central North Atlantic gave an abundance estimate of 10, 521 (95% CI 3,700-24,600) in 1995, and 14,662 (95% CI 9,400-29,900) in 2001, mainly to the east and north of Iceland but also to the west (Paxton *et al.* 2009). In the Barents and Norwegian Seas, the Norwegian survey estimate was 1,059 (CV=0.25) in 1995, and 1,450 (CV=0.29) in 1996-2001 (Øien 2009). Recent survey estimates suggest that the North Atlantic population may be approaching 20,000 animals (Smith and Pike 2009). During SCANS-II and CODA surveys, numbers observed in west European waters were too low to derive abundance estimates.

Sightings from around the British Isles and Ireland have increased markedly since the early 1980s; occurring in three main areas – The Northern Isles south to eastern England; The northern Irish Sea north to West Scotland; and the Celtic Sea between Southern Ireland, Southwest Wales and Southwest England, with a few sightings and strandings also in the southern North Sea (Evans *et al.* 2003, Camphuysen & Peet 2006, Clapham & Evans 2008].

Conclusions

The humpback whale was intensively hunted in the European Atlantic during the nineteenth and first half of the twentieth century. This is thought to have caused major reductions in population size. Whaling has now largely ceased in the North Atlantic. Since the moratorium on commercial whaling in the mid 1950s, the overall population appears to be steadily increasing.

Other human pressures such as bycatch, ship strike and noise disturbance exist but may not be of a magnitude to have population level effects. Changes in prey abundance and distribution as a result of overfishing or the effects of climate change may also have an impact.

Step 2.1 - FRP assessment

As a result of many years of intense human exploitation, population levels for humpback whales in the European Atlantic will have been much lower at the time when the Habitats Directive came into force than historically before exploitation, and so Favourable Reference Population size should exceed Current Value, although by what magnitude is very difficult to say with confidence. Genetic analyses at the mtDNA control region suggests that FRP could be two or three times the Current Value, although this is based partly on what the generation length is set at (Roman & Palumbi 2003).

Whole genomic analysis using restriction site associated DNA sequencing is recommended for insights into the extent to which present day populations have experienced contractions in size and loss of genetic diversity. Genetic analysis enables one to estimate the effective population size (Ne) for management units prior to major human impacts.

Conclusions

Kade The order of magnitude of FRP in the North Atlantic is likely to have been in the range of 100,000-250,000 humpback whales, based upon genetic evidence. The bulk of the North Atlantic population breeds in the West Indies, some of which then migrate into northern European waters around Iceland and northern Norway. The declines that have occurred historically could continue to be reversed if conditions allow.

FRP >> CV (European Atlantic, MATL and MMAC).

Step 2.2 - FRR assessment

The range of the humpback whale in Europe does not appear to have changed much from what it was during the period of commercial exploitation, although by the latter half of the twentieth century, humpbacks had become very scarce in the vicinity of the British Isles (Evans *et al.* 2003). There is no historical evidence to suggest the species was ever common in the Mediterranean or the Baltic. That FRR > CV relates to it being regular in its former range in areas where it is currently only an occasional visitor.

Conclusions

FRR > CV (European Atlantic, MATL and MMAC).

References

Alter, S.E. and Palumbi, S.R. (2009) Comparing evolutionary patterns and variability in the mitochondrial control region and cytochrome B in three species of baleen whales. *Journal of Molecular Evolution*, 68: 97–111.

Berrow, S., Whooley, P., O'Connell, M., and Wall, D. (2010) *Irish Cetacean Review (2000-2009*). Irish Whale and Dolphin Group, Kilrush. 60pp.

Camphuysen, C.J. and Peet, G. (2006) *Whales and dolphins in the North Sea*. Fontaine Uitgevers, Kortenhoef, The Netherlands.

Cerchio, S., Strindberg, S., Vollins, T., Bennett, C., and Rosenbaum, H. (2014) Seismic surveys negatively affect humpback whale singing activity off Angola, *PLoS One*, 2014;9(3): e86464.

Charif, R.A., Clapham, P.J., and Clark, CW. (2001) Acoustic detections of singing humpback whales in deep waters off the British Isles. *Marine Mammal Science*, 17: 751-768.

Clapham, P.J. (1996) The social and reproductive biology of humpback whales: an ecological perspective. *Mammal Review*, 26: 27-49.

Clapham, P.J. (2009) Humpback Whale *Megaptera novaeangliae*. In: *Encyclopedia of Marine Mammals*. Perrin, W.F., Wursig, B. and Thewissen, J.G.M. (eds). Academic Press, London. 2nd edition, pp. 582-585.

Clapham, P. and Evans, P.G.H. (2008) Humpback whale *Megaptera novaeangliae.* Pp. 663-665. In: *Mammals of the British Isles*. (Eds. S. Harris and D.W. Yalden). Handbook. 4th Edition. The Mammal Society, Southampton. 800pp.

Clapham, P.J. and Mead, J.C. (1999) *Megaptera novaeangliae*. *Mammalian Species*, 604: 1-9.

CODA (2009) Cetacean Offshore Distribution and Abundance in the European Atlantic (CODA), 43pp.

Collett, R. (1912) Norges Pattedyr (Norges Hvirveldyr I). H. Aschehoug, W. Nygaard, Kristiania (Oslo).

Evans, P.G.H. and Bjørge, A. (2013) Impacts of climate change on marine mammals. Marine Climate Change Impacts Partnership (MCCIP) Science Review 2013, 134-148. Published online 28 November 2013 doi:10.14465/2013.arc15.134-148.

Evans, P.G.H., Anderwald, P., and Baines, M.E. (2003) *UK Cetacean Status Review.* Report to English Nature and Countryside Council for Wales. Sea Watch Foundation, Oxford. 160pp.

Evans, P.G.H., Baines, M.E., and Anderwald, P. (2011) Risk Assessment of Potential Conflicts between Shipping and Cetaceans in the ASCOBANS Region. ASCOBANS AC18/Doc. 6-04 (S). 32pp.

Hammond, P.S., Berggren, P., Benke, H., Borchers, D.L., Collet, A., Heide-Jørgensen, M.P., Heimlich, S., Hiby, A.R., Leopold, M.F., and Øien, N. (2002) Abundance of harbour porpoise and other cetaceans in the North Sea and adjacent waters. *Journal of Applied Ecology*, 39: 361-376.

Hammond, P.S., Macleod, K., Berggren, P., Borchers, D.L., Burt, M.L., Cañadas, A., Desportes, G., Donovan, G.P., Gilles, A., Gillespie, D., Gordon, J., Hiby, L., Kuklik, I., Leaper, R., Lehnert, K., Leopold, M., Lovell, P., Øien, N., Paxton, C.G.M., Ridoux, V., Rogan, E., Samarra, F., Scheidat, M., Sequeira, M., Siebert, U., Skov, H., Swift, R., Tasker, M.L., Teilmann, J., Van Canneyt, O., and Vázquez, J.A. (2013) Cetacean abundance and distribution in European Atlantic shelf waters to inform conservation and management. *Biological Conservation,* 164**:** 107–122.

Hazevoet, C.J. and Wenzel, F.W. (2000) Whales and dolphins (Mammalia, Cetacea) of the Cape Verde Islands, with special reference to the Humpback Whale *Megaptera novaeangliae* (Borowski, 1781). *Contributions to Zoology*, 69: 197-211.

Holt, S.J. and Mitchell, E.D. (2004) Counting whales in the North Atlantic*. Science*, 303: 39–40.

Jann, B., Allen, J., Carrillo, M., Hanquet, S., Katona, S.K., Martin, R.R., Reeves, R.R., Seton, R., Stevick, P.T., and Wenzel, F.W. (2003) Migration of a humpback whale (*Megaptera novaeangliae*) between the Cape Verde Islands and Iceland. *Journal of Cetacean Research and Management*, 5(2): 125−129.

Katona, S.K. and Beard, J.A. (1990) Population size, migrations and feeding aggregations of the humpback whale (Megaptera novaeangliea) in the western North Atlantic Ocean. Reports of the International Whaling Commission, 295–305.

Kinze, C.C. (1995) Danish whale records 1575-1991 (Mammalia, Cetacea). Steenstrupia, 21: 155-196.

Kinze, C.C., Jensen, T., Tougaard, S., and Baagøe, H.J. (2010) Danske hvalfund i perioden 1998-2007 [Records of cetacean strandings on the Danish coastline during 1998-2007]. Flora og Fauna, 116(4): 91-99.

Laran, S., Authier, M., Blanck, A., Doremus, G., Falchetto, H., Monestiez, P., Pettex, E., Stephan, E., Van Canneyt, O., and Ridoux, V. (2017) Seasonal distribution and abundance of cetaceans within French waters: Part II: The Bay of Biscay and the English Channel. *Deep-Sea Research II*, 14: 31-40.

Lockyer, C. and Pike, D. (eds.) (2009) *North Atlantic Sightings Surveys. Counting whales in the North Atlantic 1987-200*1 NAMMCO Scientific Publications Volume 7. 244pp.

Martin, A.R., Katona, S.K., Mattila, D., Hembree, D., and Waters, T.D. (1984) Migration of humpback whales between the Caribbean and Iceland. *Journal of Mammalogy*, 65: 330-333.

Mitchell, E. and Reeves, R.R. (1983) Catch history, abundance, and present status of Northwest Atlantic humpback whales. Reports of the International Whaling Commission, 5: 153–212.

Northridge, S., Cargill, A., Coran, A., Mandleberg, L., Calderan, S., and Reid, R.J. (2010) *Entanglement of minke whales in Scottish waters: an investigation into occurrence, causes and mitigation*. Sea Mammal Research Unit, Final Report to Scottish Government CR/2007/49. 57pp.

Notarbartolo di Sciara, G. and Birkun, A. (2010) *Conserving whales, dolphins and porpoises in the Mediterranean and Black Seas.* An ACCOBAMS Status Report. ACCOBAMS, Monaco. 212pp.

Øien, N. (2009) Distribution and abundance of large whales in Norwegian and adjacent waters based on ship surveys 1995-2001. Pp. 31-47. In: *North Atlantic Sightings Surveys. Counting whales in the North Atlantic 1987-200*1 (C. Lockyer and D. Pike, eds.). NAMMCO Scientific Publications, Volume 7. 244pp.

Paxton, C.G.M., Burt, M.L., Hedley, S.L., Vikingsson, G., Gunnlaugsson, Th., and Desportes, G. (2009) Density surface fitting to estimate the abundance of humpback whales based on the NASS-95 and NASS-2001 aerial and shipboard surveys. Pp. 143-159. In: *North Atlantic Sightings Surveys. Counting whales in the North Atlantic 1987-200*1 (C. Lockyer and D. Pike, eds.). NAMMCO Scientific Publications Volume 7. 244pp.

Punt, A.E., Friday, N.A., and Smith, T.D. (2006) Reconciling data on the trends and abundance of North Atlantic humpback whales within a population modelling framework. *Journal of Cetacean Research and Management*, 8: 145–159.

Reid, J.B., Evans, P.G.H., and Northridge, S.P. (2003) *Atlas of Cetacean Distribution in North-west European Waters*. Joint Nature Conservation Committee, Peterborough. 76pp.

Reilly, S.B., Bannister, J.L., Best, P.B., Brown, M., Brownell Jr., R.L., Butterworth, D.S., Clapham, P.J., Cooke, J., Donovan, G.P., Urbán, J., and Zerbini, A.N. (2008) *Megaptera novaeangliae. The IUCN Red List of Threatened Species 2008*: e.T13006A3405371. http://dx.doi.org/10.2305/IUCN.UK.2008.RLTS.T13006A3405371.en

Reiner, F., M.E. dos Santos, M.E., and Wenzel, F.W. (1996) Cetaceans of the Cape Verde archipelago. *Marine Mammal Science*, 12: 434-443.

Risch, D., Corkeron, P.J., Ellison, W.T., and Van Parijs, S.M. (2012) Changes in Humpback Whale Song Occurrence in Response to an Acoustic Source 200 km Away. PLoS ONE 7: e29741.doi:10.1371/journal.pone.0029741

Roman, J. and Palumbi, S.R. (2003) Whales before whaling in the North Atlantic. *Science*, 301: 508–510.

Ruegg, K., Rosenbaum, H.C., Anderson, E.C., Engle, M., Rothschild, A., Baker, C.S., and Palumbi, S.R. (2013) Long-term population size of the North Atlantic humpback whale within the context of worldwide population structure. *Conservation Genetics*, 14: 103-114.

Ryan, R., Leaper, R., Evans, P.G.H., Dyke, K., Robinson, K.P., Haskins, G.N., Calderan, S., van Geel, N., Harries, O., Froud, K., Brownlow, A., and Jack, A. (2016) Entanglement: an emerging threat to humpback whales in Scottish waters. IWC SC/66b/HIM/01. 12pp.

Smith, T.D. and Pike, D.G. (2009) The enigmatic whale: the North Atlantic humpback. NAMMCO Scientific Publications, 7: 161–178.

Smith, T.D. and Reeves, R.R. (2010) Historical catches of humpback whales, *Megaptera novaeangliae*, in the North Atlantic Ocean: estimates of landings and removals. *Marine Fisheries Review*, 72: 1–43.

Smith, T.D., Allen, J., Clapham, P.J., Hammond, P.S., Katona, S., Larsen, F., Lien, J., Mattila, D., Palsbøoll, P.J., Sigurjønsson, J., Stevick, P.T., and Øien, N. (1999) An ocean-basin-wide mark-recapture study of the North Atlantic humpback whale *(Megaptera novaeangliae). Marine Mammal Science,* 15: 1-32.

Stevick, P.T., Allen, J., Clapham, P.J., Friday, N., Katona, S.K., Larsen, F., Lien, J., Mattila, D.K., Palsbøll, P.J., Sigurjónsson, J., Smith, T.D., Øien, N., and Hammond, P.S. (2003). North Atlantic humpback whale abundance and rate of increase four decades after protection from whaling. *Marine Ecology Progress Series*, 258: 263-273.

Stevick, P.T., Allen, J., Clapham, P.J., Friday, N., Katona, S.K., Larsen, F., Lien, J., Mattila, D.K., Palsbøll, P.J., Sigurjónsson, J., Smith, T.D., Øien, N., and Hammond, P.S. (2006) Population spatial structuring on the feeding grounds in North Atlantic humpback whales (*Megaptera novaeangliae*)*. Journal of Zoology, London,* 270: 244-255.

Stevick, P.T., Øien, N., and Mattila, D.K. (1998) Migration of a humpback whale (*Megaptera novaeangliae*) between Norway and the West Indies. *Marine Mammal Science*, 14: 162-165.

1.5 Minke whale (*Balaenoptera acutorostrata*) in the European Atlantic

Step 1.1 - Biology of the species

The minke whale (*Balaenoptera acutorostrata*) is the smallest of rorqual whales of the family Balaenopteridae occurring in the North Atlantic. It is usually seen singly or in groups of 2-3 individuals although it may aggregate into larger groups of tens or even hundreds of individuals when in the vicinity of prey concentrations. Minke whales frequently approach vessels and may bow- or stern-ride them. Evidence from recognisable individuals indicates some seasonal site fidelity over a small gepgraphic range. Differential migration by sex and age may lead to segregation by sex and breeding condition. Breeding is diffusely seasonal, possibly occurring mainly in wintertime. Females are thought to reach sexual maturity around 6 years, and males at 7 years of age. Once sexually mature, females give birth to a single calf annually or every 2 years. Longevity is 40-50 years although the maximum age recorded is 57.5 years. Generation length has been estimated at 13.0 years (22.1 years in populations at a stable state). The diet comprises mainly shoaling meso- and benthopelagic fish, and includes herring, sandeel, sprat, cod, haddock, saithe, whiting, mackerel, blue whiting and pouts, as well as small cephalopods. In polar regions, capelin and plankton (copepods, euphausiids) may predominate. More detailed accounts of the biology of the species can be found in Stewart & Leatherwood (1985), Horwood (1990), Anderwald *et al.* (2008), and Perrin & Brownell (2009).

Figure 1.5.1 *North Atlantic Distribution of Minke Whale (depicting those areas where the species is thought to regularly occur)*

Step 1.2 - Spatial scale of functioning

The species has a cosmopolitan distribution from the tropics to the ice edge in both hemispheres, though more uncommon in equatorial waters (Figure 1.5.1). It is widespread along the Atlantic seaboard of Europe from Norway south to the southern tip of Portugal, as well as in the North Sea, although abundance is greatest in the north (Reid *et al.* 2003, CODA 2009, Hammond *et al.* 2013). Around the British Isles and Ireland, highest numbers occur off the north and west coasts of Scotland and the Hebrides, the west and south coasts of Ireland, central part of the Irish Sea including the Celtic Deep, and in the northern and central North Sea including around the Dogger Bank; it is rare in the southernmost North Sea and eastern half of the English Channel (Evans *et al.* 2003, Reid *et al.* 2003, Camphuysen & Peet 2006, Robinson *et al.* 2009, De Boer 2010, Berrow *et al.* 2010, Anderwald *et al.* 2011, Baines & Evans 2012, Hammond *et al.* 2013). In the western English Channel south to southwest Portugal, it is present but uncommon out to the edge of the continental shelf, but is largely absent from the deeper parts of the Bay of Biscay (Evans *et al.* 2003, Reid *et al.* 2003, CODA 2009, Hammond *et al.* 2013). The species is a casual visitor to Inner Danish waters, the Baltic and Mediterranean (mainly the western part) (Kinze *et al.* 2010, Notarbartolo di Sciara & Birkun 2010).

Most of the occasional strandings and sightings in the Mediterranean occurred in the Algero-Provençal and Tyrrhenian subregions, although rare occurrences are known from the Levantine Sea off Israel.

Minke whales from high latitudes are thought to migrate southwards to winter in lower latitudes (Risch *et al.* 2014); at mid-latitudes, however, such as around the British Isles and Ireland, at least some have been recorded in every month of the year (Anderwald & Evans 2007).

There is some genetic evidence for two sympatric stocks existing in the North Atlantic, with overlapping ranges, but otherwise no evidence of population structure (Anderwald *et al.* 2011). The implication is that minke whales range extensively across the North Atlantic seasonally, but segregate to some extent on at least two breeding grounds (as yet unidentified).

The population category should be largely the same as humpback whale (MNR4) except that reproduction most likely can occur within the region, and movements may not necessarily be directional – they may simply disperse for the most part off the continental shelf in winter. Although there is some summer site fidelity, annual home ranges almost certainly exceed MS level.

Conclusions

Current knowledge indicates that the minke whale in the Atlantic and North Sea, which is its main range in Europe, should be listed either under category MR4, a population of a migratory species with individuals with large home range and showing large cyclic, directed movements. The same population can occur in different Member States in time. It is not known whether migratory behaviour results in non-reproductive populations in Europe.

Step 1.3 - Historical perspective: what happened to the species?

Whaling for minke whales in the North Atlantic dates back to at least the Middle Ages when a fishery existed near Bergen, Norway (Collett 1912). However, it was not until the 1920’s that it became a target of commercial whaling when stocks of the larger baleen whales like blue, fin and sei whale had become so depleted that it was no longer economic to target them. Exploitation was greatest between the 1940s and 1980s, with recorded catches totalling about 140,000 (Mitchell 1975, Reilly *et al.* 2008). The largest catches were by Norwegian “small-type” whalers who have taken about 120,000 since 1948, mainly in the Northeast Atlantic. Annual catches peaked at over 4,000 in the late 1950s, declining to about 2,000 annually in the early 1980s. Whaling in Iceland began in 1914 and between then and 1980, approximately 3,362 minke whales were taken (Sigurjonsson 1980). Catches were phased out from 1984 to 1987. Commercial minke whaling resumed in 1993 at a lower level and continues to the present.

Calculations of pre-exploitation population sizes have been fraught with difficulties. The often widely different estimates obtained may be due to: inaccuracies in the catch record, uncertainties surrounding genetic estimates, and/or differences in time scales applied to the estimates (Roman & Palumbi 2003, Holt & Mitchell 2004, Punt *et al.* 2006, Alter & Palumbi 2009, Smith & Reeves 2010, Ruegg *et al.* 2013). However, they generally indicate marked reductions in population sizes for most of the baleen whale species since commercial whaling started. Based upon mtDNA analyses of genetic diversity, for example, the pre-exploitation population size of North Atlantic minke whales has been estimated at 265,000 compared with a “current” estimate of 149,000 (Roman & Palumbi 2003). Since the moratorium on commercial whaling was imposed by the IWC in 1986, whaling has largely ceased and populations of a number of species including minke whale appear to be recovering. The most recent estimate for the entire North Atlantic was 182,000 (Reilly *et al.* 2008).

The minke whale is still exploited in Atlantic Europe by Norway, under objection of the IWC, resulting in c. 24,300 animals taken since 1978 (averaging 200-600 per year). However, there is no indication of a decline in numbers of minke whales in NW European seas since the mid 1980s, and indeed there may have been an increase in some areas (Schweder *et al.* 1997, Evans *et al.* 2003, Skaug *et al.* 2004, Bøthun *et al.* 2009, Hammond *et al.* 2013, Paxton *et al.* 2016).

The other known causes of mortality are bycatch, mainly through entanglement in ghost netting or creel lines (Northridge *et al.* 2010) and ship strikes (Evans *et al.* 2011), although the species may be affected by noise disturbance from seismic surveys and the use of mid-frequency active sonar (Kvadsheim *et al.* 2017). Prey depletion through overfishing or the effects of climate change may also have an influence on minke whale abundance and distribution (Evans & Bjørge 2013).

Step 1.4 - Analysis of distribution and trends.
Population estimates for minke whales in the Atlantic from southern Norway southwards including the North Sea (i.e. EU range states) are from the SCANS, SCANS-II and CODA, and SCANS-III surveys. In July 1994, the SCANS survey of the North Sea, English Channel and Celtic Sea estimated 8,450 individuals (95% CI: 5,000-13,500) (Hammond *et al.*, 2002). A more extensive line transect survey (SCANS-II) over the North West European continental shelf in July 2005 gave an overall estimate of 18,958 (CV=0.35; 95% CI: 9,800-36,700) (Hammond *et al.* 2013). And the offshore CODA survey in 2007 yielded a population estimate of 6,765 (CV=0.99; 95% CI: 1,300-34,200) (CODA 2009). This latter estimate has very wide confidence intervals and was uncorrected for animals missed along the track-line, and is therefore negatively biased. For the equivalent area surveyed in 1994 and 2005, there was no evidence for a significant change in numbers (Hammond *et al.* 2013). The SCANS-III survey undertaken in July 2016 yielded an abundance estimate of 13,101 (CV=0.35; 95% CI: 7,050-26,700) (Hammond *et al.* 2017). This estimate applies to West Norwegian coastal waters, North Sea (including English Channel), West Scotland, Irish Sea, Celtic Sea, Bay of Biscay, and Iberian Peninsula, but excludes west and south of Ireland and offshore Portugal.

A population estimate for the entire North Eastern North Atlantic (based upon data from 2008-2013) gave 90,000 individuals (95% CI: 60,000-130,000) (IWC website: <www.iwc.int>), with an additional 50,000 (95% CI: 30,000-85,000) in the central North Atlantic (2005-2007) (see also Lockyer & Pike 2009). Previously, the stock seasonally inhabiting the Norwegian and Barents Seas was estimated at 86,700 individuals (95% CI: 61,000-117,000) (Schweder *et al.* 1997). Assessing minke whale numbers is difficult and controversial, since the species is inconspicuous at sea, and often reacts to survey vessels. Nevertheless, the current population appears to be in a relatively healthy state.

Conclusions

The minke whale was intensively hunted in the European Atlantic throughout much of the twentieth century. This is thought to have caused major reductions in population size by the 1980s. Whaling continues but at a much lower level, and may no longer be limiting population growth. Other human pressures such as bycatch, ship strike and noise disturbance exist but may not be of a magnitude to have population level effects. Changes in prey abundance and distribution as a result of overfishing or the effects of climate change may also have an impact.

Since the moratorium on commercial whaling in the mid 1980s, numbers taken (under objection of IWC) have been in the order of 200-600 per annum, and the overall population may now be stable or increasing.

Step 2.1 - FRP assessment
As a result of many years of intense human exploitation, population levels for minke whales in the European Atlantic will have been much lower at the time when the Habitats Directive came into force than historically, and so Favourable Reference Population size should exceed Current Value, although by what magnitude remains difficult to say with confidence. Genetic analyses at the mtDNA control region suggests that FRP could be as much as twice the Current Value (Roman & Palumbi 2003).

Whole genomic analysis using restriction site associated DNA sequencing is recommended for insights into the extent to which present day populations have experienced contractions in size and loss of genetic diversity. Genetic analysis enables one to estimate the effective population size (Ne) for management units prior to major human impacts.

The order of magnitude of FRP in the North Atlantic is likely to be in the range of 200,000-300,000 minke whales, of which maybe 60% occur in the European Atlantic. The species has a more or less continuous distribution which in summer is concentrated in the northern North Atlantic (where females may constitute c. 70% or more) (Horwood 1987, Anderwald *et al.* 2011). The declines that have occurred historically could be reversed if conditions allowed.

Conclusions

FRP > CV (European Atlantic, MATL and MMAC).

Step 2.2 - FRR assessment

The range of the minke whale in Europe does not appear to have changed much from what it was during the period of commercial exploitation, the whaling grounds being the same areas as those occupied today (Mitchell 1975, Horwood 1987). There is no historical evidence to suggest the species was ever common in the Mediterranean or the Baltic.

Conclusions

FRR = CV (European Atlantic, MATL and MMAC).

References

Alter, S.E. and Palumbi, S.R. (2009) Comparing evolutionary patterns and variability in the mitochondrial control region and cytochrome B in three species of baleen whales. *Journal of Molecular Evolution*, 68: 97–111.

Anderwald, P. and Evans, P.G.H. (2007) Minke whale populations in the North-Atlantic – an overview with special reference to UK Waters. In: An Integrated Approach to Non-lethal Research on Minke Whales in European Waters (Editors K.P. Robinson, P.T. Stevick and C.D. MacLeod). European Cetacean Society Special Publication Series, 4: 8-13.

Anderwald, P., Daníelsdottir, A.K., Haug, T., Larsen, F., Lesage, V., Reid, R.J., Vikingsson, G., and Hoelzel, A.R. (2011) Possible cryptic stock structure for minke whales in the North Atlantic: Implications for conservation and management. Biological Conservation, 144: 2479-2489.

Anderwald, P., Evans, P.G.H., Dyer, R., Dale, A., Wright, P.J., and Hoelzel, A.R. (2012) Spatial scale and environmental determinants in minke whale habitat use and foraging. *Marine Ecology Progress Series*, 450: 259-274.

Anderwald, P., Evans, P.G.H., Hoelzel, A.R., and Papastavrou, V. (2008) Minke whale *Balaenoptera acutorostrata*. Pp.665-669. In: *Mammals of the British Isles*. (Eds. S. Harris and D.W. Yalden). Handbook. 4th Edition. The Mammal Society, Southampton. 800pp.

Baines, M.E. and Evans, P.G.H. (2012) *Atlas of the Marine Mammals of Wales.* 2nd Edition. CCW Monitoring Report No. 68. 143pp.

Berrow, S., Whooley, P., O'Connell, M., and Wall, D. (2010) *Irish Cetacean Review (2000-2009*). Irish Whale and Dolphin Group, Kilrush. 60pp.

Bøthun, G., Skaug, H.J., and Øien, N., (2009) Abundance of minke whales in the northeast Atlantic based on survey data collected over the period 2002–2007. Paper SC/61/RMP2 Submitted to the Scientific Committee of the International Whaling Commission. IWC, The Red House, 135 Station Road, Impington, Cambridge CB24 9NP, UK.

Camphuysen, C.J. and Peet, G. (2006) *Whales and dolphins in the North Sea*. Fontaine Uitgevers, Kortenhoef, The Netherlands.

CODA (2009) Cetacean Offshore Distribution and Abundance in the European Atlantic (CODA), 43pp.

Collett, R. (1912) Norges Pattedyr (Norges Hvirveldyr I). H. Aschehoug, W. Nygaard, Kristiania (Oslo).

De Boer, M.N. (2010) Spring distribution and density of minke whale *Balaenoptera acutorostrata* and other marine mammals in the Central North Sea. *Marine Ecology Progress Series*, 408: 265-274.

Evans, P.G.H. and Bjørge, A. (2013) Impacts of climate change on marine mammals. Marine Climate Change Impacts Partnership (MCCIP) Science Review 2013, 134-148. Published online 28 November 2013 doi:10.14465/2013.arc15.134-148.

Evans, P.G.H., Anderwald, P., and Baines, M.E. (2003) *UK Cetacean Status Review.* Report to English Nature and Countryside Council for Wales. Sea Watch Foundation, Oxford. 160pp.

Evans, P.G.H., Baines, M.E., and Anderwald, P. (2011) Risk Assessment of Potential Conflicts between Shipping and Cetaceans in the ASCOBANS Region. ASCOBANS AC18/Doc. 6-04 (S). 32pp.

Hammond, P.S., Berggren, P., Benke, H., Borchers, D.L., Collet, A., Heide-Jørgensen, M.P., Heimlich, S., Hiby, A.R., Leopold, M.F., and Øien, N. (2002) Abundance of harbour porpoise and other cetaceans in the North Sea and adjacent waters. *Journal of Applied Ecology*, 39: 361-376.

Hammond, P.S., Macleod, K., Berggren, P., Borchers, D.L., Burt, M.L., Cañadas, A., Desportes, G., Donovan, G.P., Gilles, A., Gillespie, D., Gordon, J., Hiby, L., Kuklik, I., Leaper, R., Lehnert, K., Leopold, M., Lovell, P., Øien, N., Paxton, C.G.M., Ridoux, V., Rogan, E., Samarra, F., Scheidat, M., Sequeira, M., Siebert, U., Skov, H., Swift, R., Tasker, M.L., Teilmann, J., Van Canneyt, O., and Vázquez, J.A. (2013) Cetacean abundance and distribution in European Atlantic shelf waters to inform conservation and management. *Biological Conservation,* 164**:** 107–122.

Holt, S.J. and Mitchell, E.D. (2004) Counting whales in the North Atlantic*. Science*, 303: 39–40.

Horwood, J. (1990) *Biology and exploitation of the minke whale*. CRC Press, London.

Kinze, C.C. (1995) Danish whale records 1575-1991 (Mammalia, Cetacea). *Steenstrupia*, 21: 155-196.

Kinze, C.C., Jensen, T., Tougaard, S., and Baagøe, H.J. (2010) Danske hvalfund i perioden 1998-2007 [Records of cetacean strandings on the Danish coastline during 1998-2007]. *Flora og Fauna*, 116(4): 91-99.

Kvadsheim, P.H., DeRuiter, S., Sivle, L.D., Goldbogen, J., Roland-Hansen, R., Miller, P.J.O., Lam, F.-P.A., Calambokidis, J., Friedlaender, A., Visser, F., Tyack, P.L., Kleivane, L., and Southall, B. (2017) Avoidance responses of minke whales to 1–4 kHz naval sonar *Marine Pollution Bulletin* (In press) DOI: 10.1016/j.marpolbul.2017.05.037

Laran, S., Authier, M., Blanck, A., Doremus, G., Falchetto, H., Monestiez, P., Pettex, E., Stephan, E., Van Canneyt, O., and Ridoux, V. (2017) Seasonal distribution and abundance of cetaceans within French waters: Part II: The Bay of Biscay and the English Channel. *Deep-Sea Research II*, 14: 31-40.

Lockyer, C. and Pike, D. (eds.) (2009) *North Atlantic Sightings Surveys. Counting whales in the North Atlantic 1987-200*1 NAMMCO Scientific Publications Volume 7. 244pp.

Mitchell, E.D. (1975) Porpoise, dolphin, and small whale fisheries of the world: status and problems. *IUCN Monographs*, 3: 31-42.

Northridge, S., Cargill, A., Coran, A., Mandleberg, L., Calderan, S., and Reid, R.J. (2010) *Entanglement of minke whales in Scottish waters: an investigation into occurrence, causes and mitigation*. Sea Mammal Research Unit, Final Report to Scottish Government CR/2007/49. 57pp.

Notarbartolo di Sciara, G. and Birkun, A. (2010) *Conserving whales, dolphins and porpoises in the Mediterranean and Black Seas.* An ACCOBAMS Status Report. ACCOBAMS, Monaco. 212pp.

Paxton, C.G.M., Scott-Hayward, L., Mackenzie, M., Rexstad, E. and Thomas, L. (2016) Revised Phase III Data Analysis of Joint Cetacean Protocol Data Resources. JNCC Report No. 517. 196pp.

Perrin, W.F. and Brownell, R.L. (2009) Minke whales *Balaenoptera acutorostrata* and *B. bonaerensis*. In: *Encyclopedia of Marine Mammals*. Perrin, W.F., Wursig, B. and Thewissen, J.G.M. (eds). Academic Press, London. 2nd edition, pp. 733–735.

Punt, A.E., Friday, N.A., and Smith, T.D. (2006) Reconciling data on the trends and abundance of North Atlantic humpback whales within a population modelling framework. *Journal of Cetacean Research and Management*, 8: 145–159.

Reid, J.B., Evans, P.G.H., and Northridge, S.P. (2003) *Atlas of Cetacean Distribution in North-west European Waters*. Joint Nature Conservation Committee, Peterborough. 76pp.

Reilly, S.B., Bannister, J.L., Best, P.B., Brown, M., Brownell Jr., R.L., Butterworth, D.S., Clapham, P.J., Cooke, J., Donovan, G.P., Urbán, J. and Zerbini, A.N. (2008) *Balaenoptera acutorostrata. The IUCN Red List of Threatened Species 2008*: e.T2474A9444043. http://dx.doi.org/10.2305/IUCN.UK.2008.RLTS.T2474A9444043.en

Risch, D., Manuel Castellote, C., Clark, C.W., Davis, G.E., Dugan, P.J., Hodge, L.E.W., Kumar, A., Lucke, K., Mellinger, D.K., Nieukirk, S.L., Popescu, C.M., Ramp, C., Read, A.J., Rice, A.N., Silva, M.A., Siebert, U., Stafford, K.M., Verdaat, H., and Van Parijs, S.M. (2014) Seasonal migrations of North Atlantic minke whales: novel insights from large-scale passive acoustic monitoring networks. *Movement Ecology*, 2: 24. doi:10.1186/s40462-014-0024-3.

Robinson, K.P., Tetley, M.J., and Mitchelson-Jacob, E.G. (2009) The distribution and habitat preference of coastally occurring minke whales (*Balaenoptera acutorostrata*) in north-east Scotland. *Journal of Coastal Conservation*, 13(1): 39-48.

Roman, J. and Palumbi, S.R. (2003) Whales before whaling in the North Atlantic. *Science*, 301: 508–510.

Ruegg, K., Rosenbaum, H.C., Anderson, E.C., Engle, M., Rothschild, A., Baker, C.S., and Palumbi, S.R. (2013) Long-term population size of the North Atlantic humpback whale within the context of worldwide population structure. *Conservation Genetics*, 14: 103-114.

Schweder, T., Skaug, H.J., Dimakos, X.K., Langas, M., and Øien, N., (1997) Abundance of northeastern Atlantic minke whales, estimates for 1989 and 1995. *Reports of the International Whaling Commission*. 47: 453–483.

Sigurjonnson, J. (1982) Icelandic minke whaling 1914-1980. *Reports of the international Whaling Commission,* 32: 287-295.

Skaug, H.J., Øien, N., Schweder, T., and Bøthun, G., (2004) Abundance of minke whales (*Balaenoptera acutorostrata*) in the Northeastern Atlantic. *Canadian Journal of Fisheries and Aquatic Science*, 61: 870-886.

Smith, T.D. and Reeves, R.R. (2010) Historical catches of humpback whales, *Megaptera novaeangliae*, in the North Atlantic Ocean: estimates of landings and removals. *Marine Fisheries Review*, 72: 1–43.

Stewart, B.S. and Leatherwood, S. (1985) Minke whale *Balaenoptera acutorostrata*. In: S. H. Ridgway and R. Harrison (eds), *Handbook of marine mammals, Vol. 3: The Sirenians and Baleen Whales*. Academic Press., London, pp. 91-136.

2 Seals and turtles

Susan Gubbay

2.1 Grey seal (*Halichoerus grypus*) in the North-east Atlantic and in the Baltic sea

Step 1.1 - Biology of the species
Grey seals (*Halichoerus grypus*) are marine mammals of the *Phocidae*, the earless or true seals. They are streamlined, excellent swimmers and divers and rely on their blubber for insulation. For much of the year they alternate foraging at sea with visits haul out sites on land where they may be found in groups. Studies show that although seals may breed in the same region where they forage this is not necessarily the case (Russell *et al.,* 2013). Grey seals gather together for breeding, moulting and hauling out. They typically produce a single offspring after which there is a short lactation period when the pups feed on milk which is particular rich in fat. Males compete directly with one another for females (the grey seal is highly polygynous) and typically reach sexual maturity at 6 years but most do not become active breeders until 12-18 years of age. The males are larger than females, averaging more than 2 m but have a shorter life span, seldom surviving more than 25 years. Female grey seals may live for 35 years. Grey seals are generalist predators though largely demersal or benthic feeders, with a diet that varies by season and location (Evans & Stirling, 2001; Bowen, 2016).

Step 1.2 - Spatial scale of functioning
The Grey seal occurs in the North Atlantic, Barents and Baltic Sea. The main concentrations are found on the east coast of Canada and USA and in north-west Europe. Within Europe the vast majority of grey seals are distributed in the UK, and especially at the Orkneys, Shetland, and the Isle of May. Significant colonies also occur along the English east coast and in Wales. Other colonies are found in Ireland, Iceland, and the Faeroe Islands. The largest group of grey seals in continental Europe is found in the Dutch Wadden Sea, but they also occur in Normandy, Schleswig-Holstein, the Kattegat-Skagerrak, the Norwegian west coast and within the Baltic Sea.

Grey seals have specific breeding colonies which they return to in autumn. In general, female grey seals return to the same colony to breed in successive years and often breed at the colony in which they were born (Pomeroy *et al.,* 1994). They forage in the open sea and return regularly to haul out on land where they rest, moult and breed. They may range widely to forage and frequently travel over 100km between haul out sites. Foraging trips can last anywhere between 1-30 days (Russell *et al.,* 2013; Russell & McConnell, 2014).

There are three genetically distinct groups of grey seal; western and eastern North Atlantic groups, which are believed to have diverged approximately 1 – 1.2 million years ago; and a Baltic Sea group which is believed to have diverged from the North-East Atlantic population less than 9,000 years ago (Boskovic *et al.,* 1996; Decker *et al.,* 2017; Härkönen *et al.,* 2007; Olsen *et al.,* 2016). There is a degree of reproductive isolation between grey seals, for example between those that breed around Scotland and south-west England, although satellite tracking studies reveal that there is intermingling. Grey seals in Ireland, for example are generally considered part of a larger interacting population with at least some groups of seals in the UK and France (Ó Cadhla *et al.,* 2013) and the same between grey seals in the UK and the Wadden Sea (Brasseur *et al.,* 2015). There is also some hybridization between Baltic Sea and north east Atlantic sub-species with the latter recolonizing historically important areas for the Baltic Sea sub-species in the Kattegat and Skagerrak (Fietz *et al.,* 2016). The strength of the linkage depends upon the distance between the sites and the density of animals at each site. Individual grey seals based at a specific haul out site often make repeated trips to the same region offshore, but will occasionally move to a new haul out site and begin foraging in a new region.

Grey seals colonies are part of a metapopulation across several Member States rather than having discrete populations within individual Member States (Harrison *et al.,* 2006: Härkönen *et al.,* 2007). In setting and reporting on Ecological Quality Objectives (EcoQOs) for the grey seal in the North Sea, OSPAR identified nine geographical areas as 'management units'; four in the UK, two in Germany, and one each to cover grey seal pupping areas in France, the Netherlands and Norway (OSPAR, 2005). Subsequently, with a change in terminology to 'assessment units' as advised by ICES the OSPAR Common Indicator Assessment for grey seals uses monitoring data from twenty-six grey seal breeding areas, albeit grouped under two larger regions, the North Sea and western Britain, Ireland and western France (Table 2.2.1; ICES, 2014). In the Baltic Sea, HELCOM use the entire Baltic Sea area as the assessment unit for grey seal (HELCOM, 2006).

Table 2.1.1 *Marine Strategy Framework Directive (MSFD) assessment units within Member States*

Country	MSFD assessment unit: Main breeding area monitored (from ICES, 2014)
United Kingdom	North Sea: Shetland, Scotland
	North Sea: Orkney, Scotland
	North Sea: Fast Castle, Isle of May and adjacent colonies, Scotland
	North Sea: Moray Firth, east Scotland
	North Sea: Farne Islands, east England
	North Sea: Donna Nook & Norfolk colonies, SE England
Netherlands	North Sea: Wadden Sea
	North Sea: Delta
Germany	North Sea: Schleswig-Holstein, Wadden Sea
	North Sea: Helgoland
Denmark	North Sea: Limfjord
	North Sea: Kattegat
Norway	North Sea: Rogaland
France	North Sea: Archipelago of Sept Iles and adjacent haul-outs
	North Sea: Archipelago of Molene and adjacent haul-outs
	North Sea: Baie de Somme and adjacent haul-outs
United Kingdom	Celtic Sea: West Scotland
	Celtic Sea: Western Isles, Scotland
	Celtic Sea: Welsh coasts and southwest England
Ireland	Celtic Sea: Sturral (near Glen head) to Maghera in SW Co.Donegal
	Celtic Sea: the Inishkea Island group off northwest Co.Mayo
	Celtic Sea: Inishshark, Inishgort and associated islands off NW Co.Galway
	Celtic Sea: Islands around Slyne Head, Co.Galway
	Celtic Sea: the Blasket Islands, Co.Kerry
	Celtic Sea: the Slatee Islands, Co.Wexford
	Celtic Sea: Lambay Island & Ireland's Eye, Co.Dublin

Conclusions

This species could be listed under population category MR4 as individuals show large cyclic directed movements which can show strong transboundary movements from year to year and have inherently large home ranges forming one mixing population. The most appropriate population category is considered to be MR1/S1 as the species is widespread with more or less continuous distribution (often crossing national boundaries) and populations (assessment units) with more or less exchange at or below national level and is not generally considered to be a fully migratory species.

Regarding the scale of assessment, grey seals in the Baltic Sea should be considered a separate management unit from those in the North-East Atlantic as these are recognised as a separate sub-species. There is scope to define several other sub-populations but with no clear scientific consensus on the most appropriate boundaries, it is recommended that FRPs are based on the twenty-six grey seal breeding areas used as assessment units by OSPAR and recommended by ICES.

Step 1.3 - Historical perspective: what happened to the species?

The grey seal is known to have been more abundant and have a greater range in the past, including a continuous distribution of grey seal colonies along mainland Europe. Seal remains have been found in some of the earliest human settlements in Scotland while sub-fossil remains reveal that grey seals were common along mainland Europe. Their presence and abundance was sufficient to support the hunting of grey seal pups during the Stone Age all along the European continent as well as from more remote sites (Härkönen *et al.,* 2007).

Hunting of seals, including the grey seal, has been the main historical anthropogenic pressure on this species. Populations were severely depleted in the 16th and 17th centuries and this trend continued up to the 20th century largely because of commercial exploitation for their meat, skins and oils. Hunting pressure also resulted in its extirpation from some areas. For example, grey seals were no longer present in the Baltic Sea area of Germany and Poland by 1912, and extirpated from the Kattegat by the 1930s. During the 20th century additional pressures from human activity were the culling of seal species that were considered pests, and the effects of pollution (PCBs) which reduced fertility.

There is good historical information, based on the hunting records and bounty systems, on which to base population estimates for some seal species in some locations. These have documented collapses in populations the Baltic Sea grey seal, in the Wadden Sea, the Kattegat and the Skagerrak, for example. The last breeding populations of the grey seal disappeared in the 16th century in the Wadden Sea and before 1900 in the Kattegat-Skagerrak and the Southwestern Baltic Sea as a result of an extermination campaign. No regular pupping occurred along mainland Europe until the end of the 1970s. After hunting was banned and protected areas were designated most populations started to increase exponentially (Heide-Jørgensen & Härkönen 1988; Kokko *et al.,* 1999, Harding & Härkönen 1999; Härkönen *et al.,* 2007).

The Baltic Sea grey seal is recovering after a century of bounty hunting and three decades of low fertility rates caused by environmental pollution. Population estimates from the beginning of the 20th century suggest that numbers exceeded 90,000 with a rapid decline to about 20,000 animals in the 1940s (Figure 2.2.1). Environmental pollution by organochlorines is considered the most likely reason for continued decline in the 1960s and 1970s after hunting had ceased. In 2014 the grey seal population in the Baltic Sea was reported as around 32,000 (HELCOM, 2016).

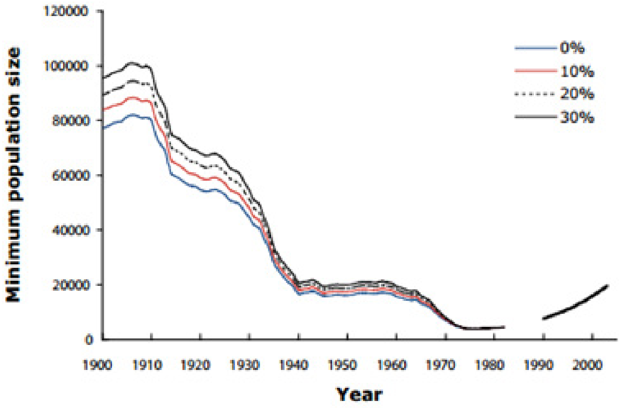


Figure 2.1.1 *Historical minimum population size of grey seals in the Baltic, estimated from hunting statistics (from Harding et al., 2007)*

Grey seal numbers in the North-East Atlantic have also been increasing in recent decades. For example, there has been a continual increase in pup production at grey seal colonies in the UK since regular surveys began in the 1960s as well as expansion of newer colonies on the mainland coastal sites on the east coast of England in recent years where grey seals have probably never previously bred in significant numbers (SCOS, 2015). In the Dutch Wadden Sea, grey seals started to recolonise coastal haul outs in the 1950s after disappearing from the area in the 16th century. A few breeding colonies were re-established in the German and Dutch parts of the Wadden Sea in the 1980s and the first confirmed sighting of a breeding grey seal (with pup) in the Danish Wadden Sea in recent times was recorded in 2014 (Jensen *et al.,* 2015).

Recent increases in number and the recolonization of parts of its historic range indicate that the grey seal is recovering naturally.

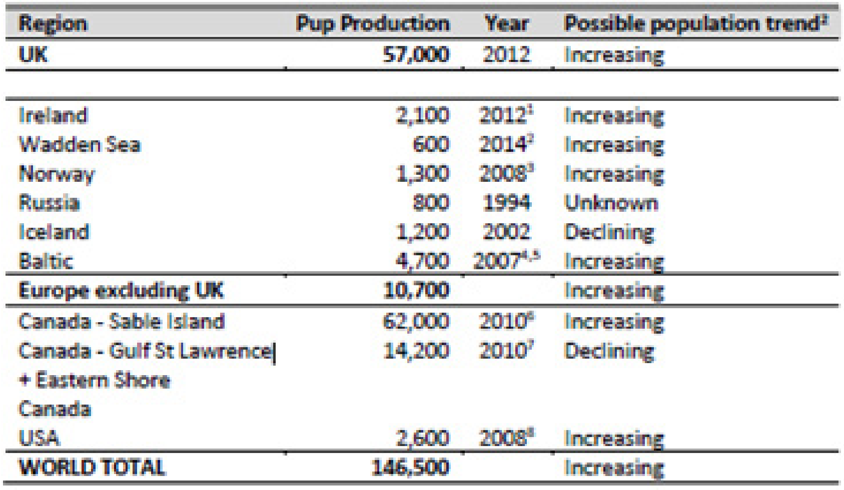
Step 1.4 - Analysis of distribution and trends

The historical distribution and abundance of the grey seal was significantly greater that the present day. There was a dramatic decline during the 16th and 17th centuries and continued decline in the early part of the 20th century. Based on hunting records the scale of the decline exceeded 90% in some areas (Kattegat and Skagerrak and Baltic Sea) and by 100% in the Dutch Wadden Sea where it was locally extinct by the 16th century. Reproducing grey seal colonies were also absent along mainland Europe coasts until the 1970s (e.g. Brasseur *et al.,* 2015; Harding *et al.*, 2007; SCOC, 2015).

In the Baltic Sea, estimates of population size at the beginning of the 20th century vary from 80,000 – 100,000 individuals. In the 1940s, with high hunting pressure there was a sharp decline to about 20,000 seals and in the 1970s there were probably between 2,000-3,000 Baltic Sea grey seals. In the Baltic Sea, the grey seal was therefore considered to have been depleted to about 5- 10% of pristine abundances before protective measures were taken. Population growth exceeded 10% per year between the early 1990s and mid-2000s, decreased to around 6% between 2010-12 and is now increasing again (e.g. Heide-Jørgensen & Härkönen 1988; Kokko *et al.,* 1999; Harding & Härkönen 1999; Harding *et al.* 2007; Kauhala *et al.,* 2012; Härkönen *et al.,* 2013).

Short term trends in grey seal populations are based on a combination of pupping data and counts of adult seas. The overall trend is one of increasing numbers (Table 2.1.2). In the UK, for example, the average increase in grey seal pup production between 2006 to 2012 ranges from 15.1% for colonies at Donna Nook & East Anglia, 11.6% in the Firth of Forth, 3.3% in the Outer Hebrides and 3.0% in Orkney (Figure 2.1.2; SCOS, 2015). In the Netherlands pup numbers increased at a rate of 19% per year between 1985 and 2010 (Figure 2.1.3). In the Baltic Sea, based on data from the Swedish coastline over the period 1990-2003, the mean annual rate of increase of the grey seal population is estimated to have been 7.9% over the period 2000-2014 (Figure 2.1.4; HELCOM, 2016).

Table 2.1.2 *Relative sizes and status of grey seal populations (from SCOS, 2015)*



Region	Pup Production	Year	Possible population trend[2]
UK	**57,000**	2012	Increasing
Ireland	2,100	2012[1]	Increasing
Wadden Sea	600	2014[2]	Increasing
Norway	1,300	2008[3]	Increasing
Russia	800	1994	Unknown
Iceland	1,200	2002	Declining
Baltic	4,700	2007[4,5]	Increasing
Europe excluding UK	**10,700**		Increasing
Canada - Sable Island	62,000	2010[6]	Increasing
Canada - Gulf St Lawrence + Eastern Shore Canada	14,200	2010[7]	Declining
USA	2,600	2008[8]	Increasing
WORLD TOTAL	**146,500**		Increasing

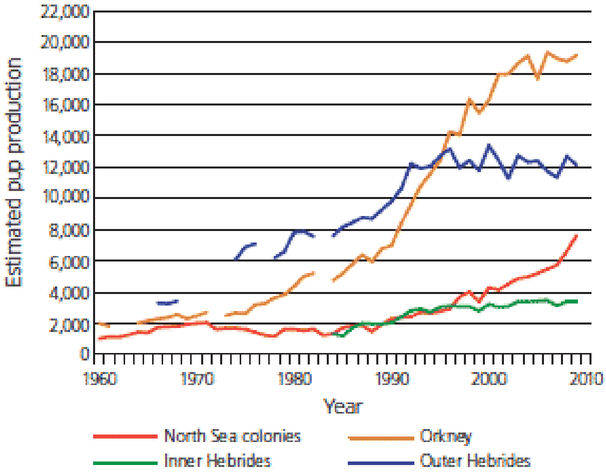


Figure 2.1.2 *UK grey seal pup production at annually monitored breeding colonies. Source. Scotland's Marine Atlas. Sea Mammal Research Unit http://www.gov.scot/Publications/2011/03/16182005/54*

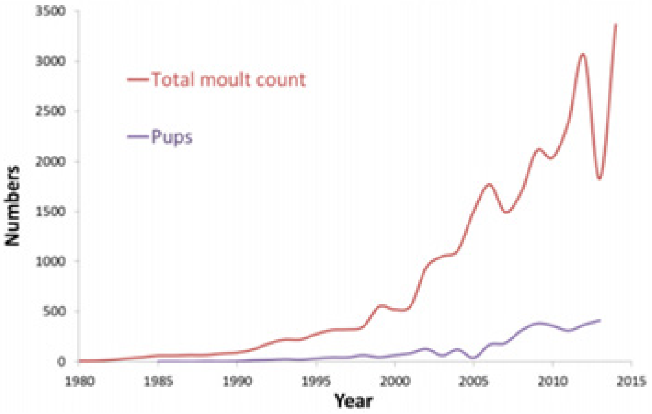


Figure 2.1.3 *Number of grey seals counted during the peak of moult each year and numbers of pups counted in the Dutch Wadden Sea between 1980 and 2014 (from Brasseur et al. 2015)*

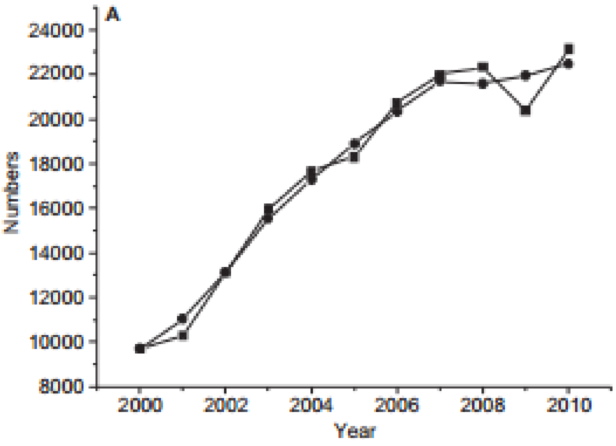


Figure 2.1.4 *Numbers of grey seals seen during sea monitoring counts (aerial surveys) in May/June in the Baltic Sea (from Kauhala et al., 2012 based on Kunnasranta, 2010)*

There have been changes in the spatial abundance of the grey seal in Europe in the recent past as well as the historical past. In the early part of the 20th century grey seal numbers were much lower around

Britain, Ireland and the Wadden Sea relative to the Baltic Sea. By the 1970s when overall numbers were starting to increase, the British population made up a much higher proportion than the Baltic population (Figure 2.1.5). More recently, surveys show increasing numbers of recolonizers at historic haul out sites, for example in the Kattegat and the southwest Baltic although, in such areas, this has been by the North East Atlantic sub-species rather than the previously present Baltic sub-species (Fietz *et al.,* 2016).

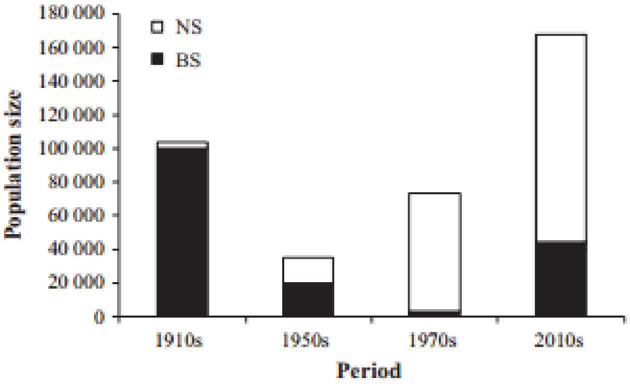


Figure 2.1.5 *Historic and contemporary abundance of grey seal in northern Europe, showing a shift in the relative population sizes of grey seals in the North Sea (NS) and the Baltic Sea (BS) (from Fietz et al., 2016)*

Conclusions

The historical distribution and abundance of this species was significantly greater that the present day. There was a dramatic decline during the 16th and 17th centuries and continued decline in the early part of the 20th century. This was principally due to hunting but other causes were the culling of seal species that were considered pests, and the effects of pollution (PCBs) which reduced their fertility. Hunting records indicate a decline which exceeded 90% in some areas (Kattegat and Skagerrak) and local extirpation. Reproducing grey seal colonies became absent along mainland Europe coasts until the 1970s. In the Baltic Sea, the grey seal is considered to have been depleted to about 5- 10% of pristine abundances before protective measures were taken.

Grey seal numbers have been increasing in the north-east Atlantic and the Baltic Sea since 1970s. They have also recolonised parts of their historic range since protective measures were introduced. This is therefore a species for which there is evidence for ongoing natural recovery of depleted populations.

Over the short term (12 years), both the North East Atlantic and Baltic Sea populations of grey seal have been increasing. There has also been some expansion of colonies in areas that were part of its former range. Census data are available to calculated the magnitude of short term trends and some longer term fluctuations and trends from the early to mid 20th century.

Step 2.1 - FRP assessment

The present day population of the grey seal is known to be substantially reduced from that of previous centuries. Numbers continued to decline into at least the middle to late 20th century in some locations and reproducing colonies are still absent from several regions where they were abundant two centuries ago.

When the Habitats Directive came into force (1992), the grey seal population in Europe was at the beginning of a period of recovery from a historical low. Population numbers continue to increase, albeit at different rates in different colonies. Harding *et al.,* (2007), using data on Baltic Sea grey seals, and taking account the likely variation in annual counts, estimate that it will take between 7 and 13 years to detect a 5% change in the growth rate of the population. Good Environmental Status (GES) under the MSFD is considered to be achieved for the Baltic Sea grey seal if the population abundance exceeds 10,000 animals and the population growth rate is greater or equal to 7% per year during the period of exponential growth and a decline of less than 10% per year during a 10 year period when populations are at carrying capacity HELCOM 2016).

1) In the Baltic Sea the population size at the beginning of the last century (modelled) is estimated to have been about 90,000 (HELCOM, 2016). In 2014 it was estimated at around 52,000, and continues to grow exponentially with no suggestion that it is approaching carrying capacity. It is also not below the calculated intrinsic growth rate of 10% which is considered the likely maximum in a healthy grey seal population (HELCOM, 2016). FRP is therefore greater than HDV for the Baltic Sea management unit.
2) The grey sea population in the North East Atlantic is also increasing. In the 1970s, for example there were estimated to be between 53-54,000 in the UK and 2,000 in Ireland (Bonner, 1976 and Smith, 1966 in Bonner 1979). For comparison, figures presented in Habitats Directive Article 17 reporting up to 2012, gave population size estimates of between 76,100-116,300 grey seals in the UK and between 7,284-9,365 in Ireland. The situation in suggested management units in the UK and Ireland is given in Table 2.1.3.

The grey seal is a good example of a naturally recovering species. Recognizing the considerable decline from historical abundance, the grey seal is now recolonizing its natural range. Instead of assessing a real-valued FRP, operators will be used until ongoing monitoring detects a stabilisation of population size and distribution for a sufficient long time.

Table 2.1.3 *Recovery of the grey seal in nine of the suggested management units (from Ó Cadhla, et al. 2013, SCS, 2015)*

		Pup production	
Country	MSFD assessment unit:Main breeding area monitored	**2010**	**2012**
UK	North Sea: Orkney, Scotland	20,312	22,926
	North Sea: Farne Islands, east England	1,499	1,603
		2005	**2009-12**
Ireland	Celtic Sea: Sturral (near Glen head) to Maghera in SW Co.Donegal	227	241 (2012)
	Celtic Sea: the Inishkea Island group off northwest Co.Mayo	386	526 (2012)
	Celtic Sea: Inishshark, Inishgort and associated islands off NW Co.Galway	220	416 (2011)
	Celtic Sea: Islands around Slyne Head, Co.Galway	68	104 (2011)
	Celtic Sea: the Blasket Islands, Co.Kerry	185	314 (2011)
	Celtic Sea: the Slatee Islands, Co.Wexford	163	151 (2009)
	Celtic Sea: Lambay Island & Ireland's Eye, Co.Dublin	58	77 (2009)

Conclusions

For all management units/populations at least FRP > CV, recognizing the natural recolonization of its natural range after a deep low.

For management units in the Baltic FRP >> CV.

Step 2.2 - FRR assessment

In the Baltic Sea, the breeding population was formerly distributed throughout the entire sea area. Today breeding seals are concentrated in the northern Baltic. The area south of latitude 58^{0} N only being recolonised very slowly. In the Danish Baltic and Kattegat, for example, only one haul out is consistently used (HELCOM, 2016). Outside the moulting period the seals are less aggregated and can be found throughout the Baltic.

In the North East Atlantic and particularly the coast of mainland Europe, breeding grey seals were formerly more widespread. Colonies gradually started to recolonize formerly used sites since the 1970s but they are still absent from formerly known breeding sites. Around the UK, monitoring since the 1960s, shows no change in the range where breeding colonies are distributed but now considerably more breeding colonies within their range.

Conclusions

Although the population of grey seals is increasing, the distribution and range of breeding colonies remains reduced compared to historical reports.

For all populations and management units at least FRR > CV.

For management units in the Baltic, FRR >> CV is more appropriate.

References

Bonner, W.N. 1976. The stocks of grey seals (*Halichoerus grypus*) and common seals (*Phoca vitulina*) in Great Britain. Publ.Ser.Natl.Environ.Res.Counc. (C) (16).

Bonner, W.N. 1979. Grey Seal. In; FAO (1979) Mammals in the seas. Volume II. Pinniped Species Summaries and Report on Sirenians. FAO Fisheries Series No5, Vol II. FAO, Rome 1979. Pg 90-94.

Boskovic, R., *et al.*, 1996. Geographic distribution of mitochondrial DNA haplotypes in grey seals (*Halichoerus grypus*). *Can. J. Zool*. 74: 1787-1796.

Bowen, D. 2016. *Halichoerus grypus*. The IUCN Red List of Threatened Species 2016: e.T9660A45226042. http://dx.doi.org/10.2305/IUCN.UK.2016-1.RLTS.T9660A45226042.en. Downloaded on 16 February 2017.

Brasseur, S., *et al.,* 2015. Pupping habitat of grey seals in the Dutch Wadden Sea. IMARES C009/15. IMARES Wageningen.

Brasseur S.M.J.M. *et al.,* 2015 Rapid recovery of Dutch grey seal colonies fueled by immigration. Marine Mammal Science 31:405-426.

Dekker, C. *et al.* 2017 Mitochondrial DNA reveals historical maternal lineages and postglacial expansion of the grey seal in European Waters. Mar.Ecol.Prog. Ser. DOI: 10.3354/meps12003.

Evans, P.G.H. & Stirling, I. 2001. Life History Strategies of Marine Mammals. Chapter 1, pg7-62. In Evans, P.G.H. & Raga, J.A. (Eds) Marine Mammals: Biology and Conservation. Kluwer Academic/Plenum Publishers. 629.

Fietz,K., Galatius, A., Teilman, J. *et al.,* 2016. Shift of grey seal subspecies boundaries in response to climate, culling and conservation. Molecular Ecology. 25:4097-4112.

Harding, K.C. *et al.,* 2007. Status of Baltic grey seals: Population assessment and extinction risk. NAMMCO Sci.Publ 6:33-56.

Harding, K.C. & Härkönen, T.J. 1999 Development in the Baltic grey seal *(Halichoerus grypus)* and ringed seal *(Phoca hispida)* populations during the 20th century. Ambio 28: 619-627.

Härkönen, T.J. *et al.,* 2007. Status of grey seals along mainland Europe from the Southwestern Baltic to France. NAMMCO Sci.Publ. 6:57-68.

Härkönen, T. *et al.,* 2013. Population growth rate, abundance and distribution of marine mammals. HELCOM Core Indicate of Biodiversity. HELCOM 34pp.

Harrison, P.J. *et al.,* 2006 Incorporating movement into models of grey seal population dynamics. J.Animal Ecol. 75:634-645.

Heide-Jørgensen, M.-P., Härkönen, T. 1988. Rebuilding seal stocks in the Kattegat-Skagerrak. Marine Mammal Science 4: 231-246.

HELCOM 2006. Conservation of Seals in the Baltic Sea Area. HELCOM Recommendation 27-28/2. Adpted 8 July 2006. Helsinki Commission. http://helcom.fi/Recommendations/Rec%2027-28-2.pdf

HELCOM 2016. Background document to the session on the BalticBOOST comparative pilot study on the use of MSFD and HD criteria for assessing the status of seals. Baltic Marine Environment Protection Commission. Ad Hoc Seal Expert Group. SEAL 10-2016. Submitted by Denmark and Sweden. https://portal.helcom.fi/meetings/SEAL%2010-2016-364/MeetingDocuments/10-1%20Background%20document%20to%20the%20session%20on%20the%20BalticBOOST%20comparative%20pilot%20study%20on%20the%20use%20MSFD%20and%20HD%20criteria.pdf

ICES 2014. OSPAR Request on Implementation of MSFD for Marine Mammals. Special request, Advice May 2014. https://www.ices.dk/sites/pub/Publication%20Reports/Advice/2014/Special%20Requests/OSPAR_Implementation_of_MSFD_for_marine_mammals.pdf

Jensen.L.F., Galatius, A. & Teilmann, J. 2015. First report on a newborn grey seal pup (*Halichoerus grypus*) in the Danish Wadden Sea since the 16th Century. Marine Biodiversity Records. doi:10.1017/S1755267215001062; Vol. 8; e131; 2015 Published online.

Kauhala, K., Ahola, M.P. & Kunnasranta, M. 2012. Demographic structure and mortality rate of a Balatic grey seal population at different stages of population change, judged on the basis of the hunting bag in Finland. Ann.Zool.Fennici 49:287-305.

Kokko, H., *et al.,* 1999: Backcasting population sizes of ringed and grey seals in the Baltic and Lake Saimaa during the 20th century. — *Annales Zoologici Fennici* **36:** 65–73.

Kunnasranta, M. 2010: Merihylkeet vuonna 2010. — *Riistaja kalatalous*. — *Selvityksiä* 21/2010: 21–22.

Ó Cadhla, O., *et al.* 2013. Monitoring of the breeding population of grey seals in Ireland, 209-2012. Irish Wildlife Manuals, No. 74. National Parks and Wildlife Service, Department of the Arts, Heritage and the Gaeltacht, Dublin, Ireland.

Olsen, M.T., *et al.,* 2016. The forgotten type specimen of the grey seal [*Halichoerus grypus* (Fabricius, 1791)] from the island of Amager, Denmark. Zool J. Linn. Soc. 2016. Doi:10.1111/zoj.12426.

OSPAR 2005. Background Document on the Ecological Quality Objective for Seal Population Trends in the North Sea. North Sea Pilot Project on Ecological Quality Objectives. Biodiversity Series. OSPAR Commission.

Pomeroy, P.P., *et al.,* 1994. Dispersion and site fidelity of breeding female grey seals (*Halichoerus grypus*) on North Rona, Scotland. *J. Zool. London*, 233: 429–447.

Russell, D.J.F., McConnell, B., Tompson, D. *et al.,* 2013. Uncovering the links between foraging and breeding regions in a highly mobile mammal. J.Appl.Ecol. 50(2); 499-509.

Russell, D.J.F. & McConnell B. 2014. Seal at-sea distribution, movements and behaviour. Sea Mammal Research Unit. Report to DECC. https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/346304/OESEA2_SMRU_Seal_distribution_and_behaviour.pdf

Smith, E.A. 1966. A review of the world’s grey sea population. *J.Zool.Lond*. 150:463-89.

Special Committee on Seals 2015. Scientific Advice on Matters Related to the Management of Seal Populations: 2015. Main Advice. 211pp. Sea Mammal Research Unit. http://www.smru.st-andrews.ac.uk/files/2016/08/SCOS-2015.pdf

2.2 Loggerhead turtle (*Caretta caretta*) in the Mediterranean

Step 1.1 - Biology of the species
The loggerhead turtle (*Caretta caretta)* is long-lived, widely distributed, and one of seven species of extant marine turtle. They are carnivorous, opportunistic feeders with a diet that includes molluscs, arthropods and echinoderms (Casale *et al.,* 2008).

Marine turtles exhibit a variety of life history traits and exploit different environments at different stages of their life cycle. In the case of the loggerhead turtle this is an oceanic-neritic developmental pattern (Figure 2.2.1A). After hatching, juvenile loggerhead turtles have an oceanic stage, which may last between 4-19 years, when they feed on plankton. After this, they return to shallow coastal waters where they switch to benthic feeding although there are instances where tagged adults have been reported from open ocean areas (McClellan & Read, 2003). More recent studies also suggest that apart from a relatively short period when the turtles have limited diving capacity, and therefore limited ability to feed on benthos efficiently, that they can frequent any oceanographic zone (Figure 2.2.1B).

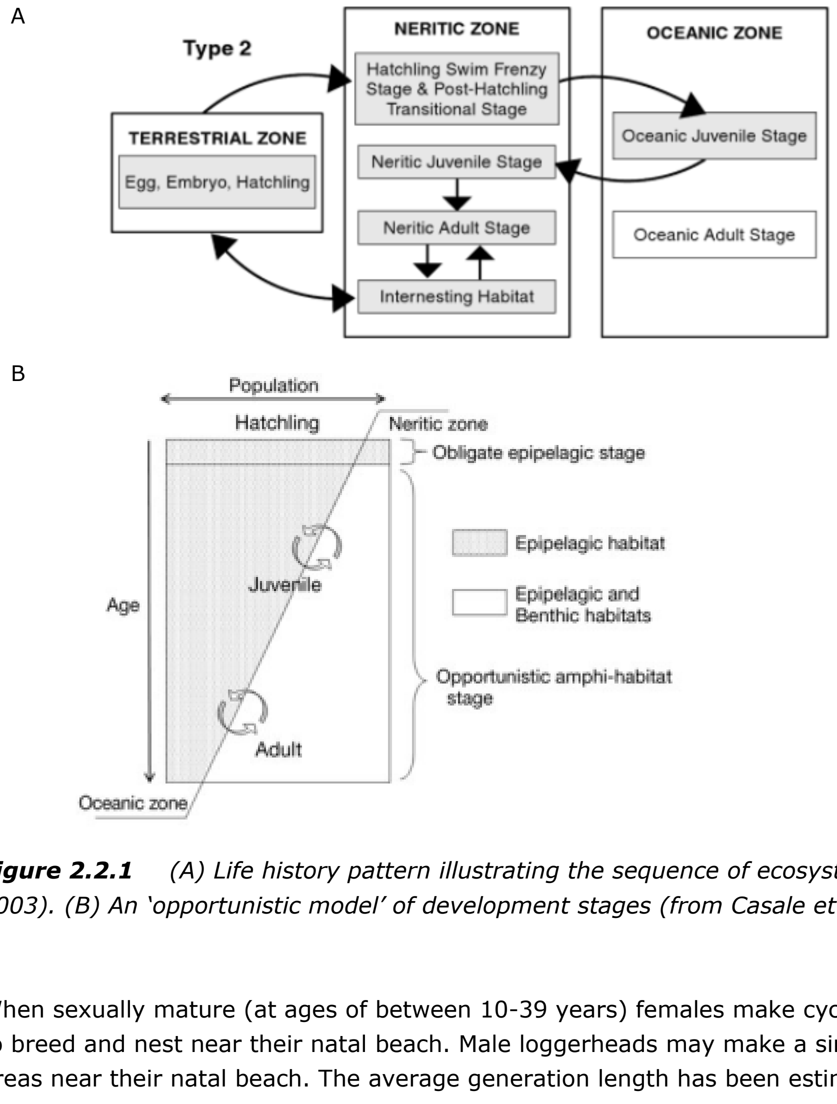


Figure 2.2.1 *(A) Life history pattern illustrating the sequence of ecosystems inhabited (from Bolten 2003). (B) An 'opportunistic model' of development stages (from Casale et al. 2008)*

When sexually mature (at ages of between 10-39 years) females make cyclic reproductive migrations to breed and nest near their natal beach. Male loggerheads may make a similar migration to breeding areas near their natal beach. The average generation length has been estimated as equal to or greater than 45 years (Avens & Snover, 2013, Bolten & Witherington 2003, Casale & Tucker, 2017). Whilst there is site fidelity of nesting, demographically independent populations mingle at foraging grounds and during migratory phases (Figure 2.2.3).

Step 1.2 - Spatial scale of functioning
The loggerhead turtle has a circumtropical distribution, being present in Mediterranean Sea, and Pacific, Indian and Atlantic Oceans (Figure 2.2.2). There are considered to be ten subpopulations and these vary widely in populations size, geographic range and population trends (Casale & Tucker, 2017).

Figure 2.2.2 *Global distribution and nesting sites for the Logger Head turtle (from Wallace et al., 2010)*

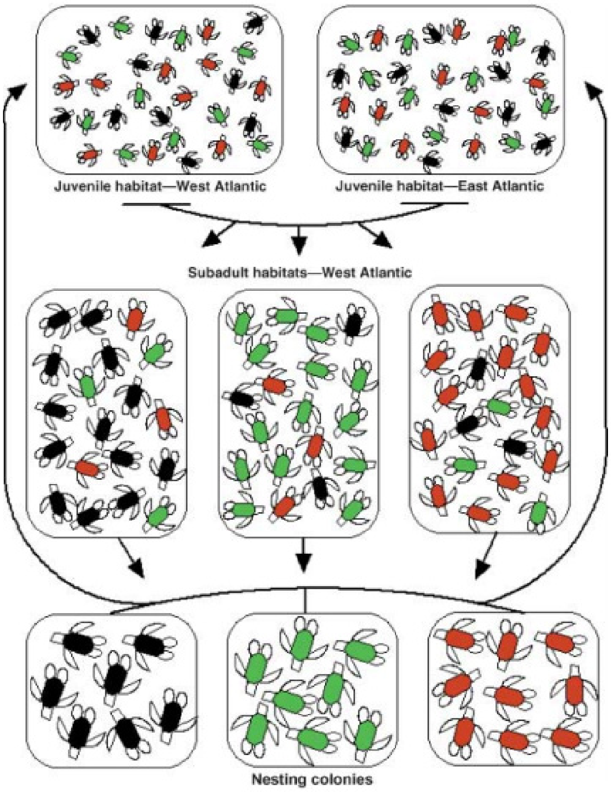


Figure 2.2.3 *Model of loggerhead population structure in the North Atlantic, using three hypothetical rookeries designated by red, green and black icons (Figure 2 from Bowen et al., 2005)*

On the feeding grounds, there are cohorts from several breeding populations as well as mixing with other regions (Figure 2.2.3). For example, there are links between nesting colonies in the North East Atlantic and the feeding populations in the North East Atlantic and the Mediterranean Sea (Bolten *et al.,* 1998, Laurent *et al.,* 1998). Tagging studies also reveal dispersal across biogeographical regions. Two examples are the post nesting movements of turtles tagged in Greece where there were marked clusters of recapture at the Gulf of Gabès and the Adriatic Sea (Margaritoulis *et al.,* 2003), and tagging of mostly juvenile turtles in an Italian project showed recaptures in both the eastern and western basins of the Mediterranean Sea.

Using information on marine turtle biogeography, including nesting sites, genetic stocks and geographic distributions, the IUCN Marine Turtle Specialist Group has developed Regional Management Units (RMUs) to provide a framework for evaluating threats and assessing the conservation status of marine turtles (Wallace *et al.,* 2010). Two RMUs have been described within Europe. A Mediterranean unit, in which 82 nesting sites, 4 mitochondrial DNA (mtDNA) stocks and 4 nuclear DNA (nDNA) stocks have been identified, and a North East Atlantic unit with 10 nesting sites, one mtDNA stock and no date on nDNA stocks (Figure 2.2.4).

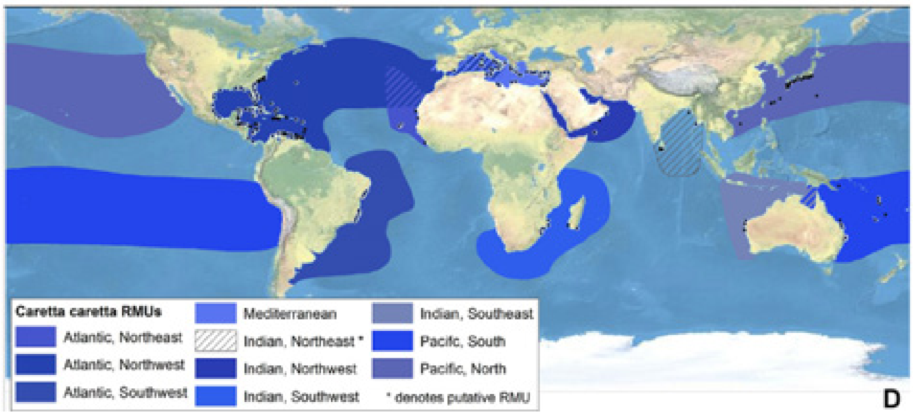


Figure 2.2.4 *Regional Management Units (RMUs) for the loggerhead turtle as proposed by IUCN (Casale, & Tucker, 2017)*

Conclusions

The loggerhead turtle (*Caretta caretta*) is a widely distributed, highly migratory species with a large home range. There is mixing of the foraging individuals at a supranational level but site fidelity of reproductive females as they return to natal beaches to lay their eggs. Non-reproductive individuals can therefore be categorized as MNR1 and the reproductive population as MR4.

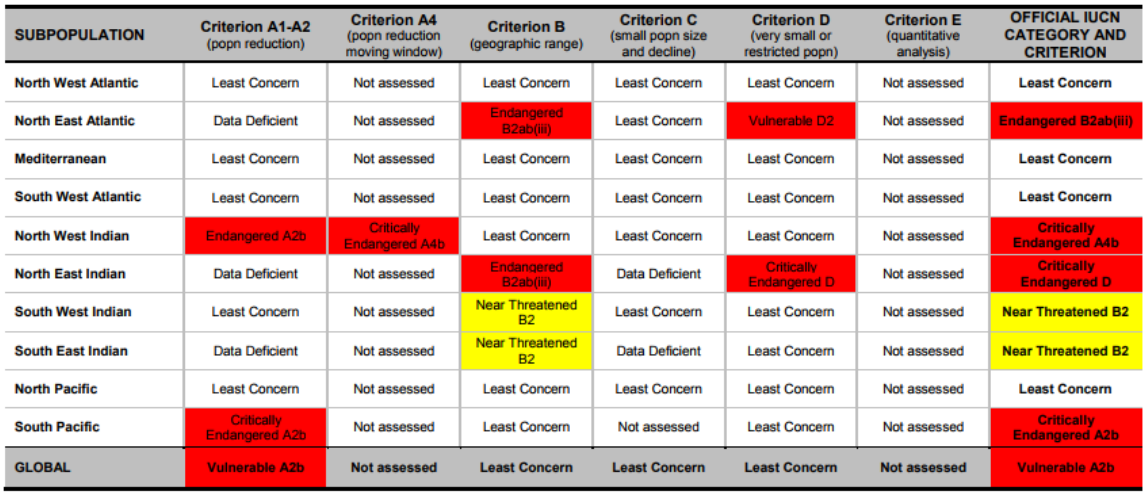
This species has a circumtropical distribution, being present in Mediterranean Sea, and Pacific, Indian and Atlantic Oceans. Ten sub-populations, which vary widely in population size, geographic range and population trends, have been identified using information on marine turtle biogeography, including nesting sites, genetic stocks and geographic distributions. Two sub-populations have been identified in Europe; a North East Atlantic and a Mediterranean sub-population. It is proposed that FRV assessment for the loggerhead turtle is carried out at a supranational level.

Step 1.3 - Historical perspective: what happened to the species?

The loggerhead turtle is exposed to different threats at different stages of its life cycle. Juveniles may be caught in pelagic fisheries such as longlining, subadults populations are affected by actions near the nesting colonies such as shrimp fisheries, dredging and pollution, and nesting females are threatened by disturbance on beaches, erosion and construction activities near their nesting sites. Turtle eggs may be taken for human consumption and there has been trade in turtle products as curios. Climate change is likely to affect hatchling sex rations while sea level rise and storm frequency may affect the condition and availability of nesting beaches.

Alongside impacts at sea there has been a degradation of nesting sites, with some areas known in the past to be nesting beaches having been lost (e.g. Malta) or severely degraded (e.g. Israel) (Margaritoulis *et al.,* 2003). Disturbance of nesting females by traffic, beachfront lighting and noise, uncontrolled development and construction are also issues. Furthermore, pollution, boat strikes near-shore fishing and use of underwater explosives have been identified as threats in the Mediterranean and elsewhere. Predators, including feral and domestic dogs, taking eggs can also be a significant added pressure on nesting beaches.

Table 2.2.1 *Summary results of subpopulation assessments and global assessments of the loggerhead turtle for all IUCN Red List Criteria, and official Red List categories and criteria. Cells shaded red and yellow indicate "Threatened" and "Near Threatened" category status respectively*



SUBPOPULATION	Criterion A1-A2 (popn reduction)	Criterion A4 (popn reduction moving window)	Criterion B (geographic range)	Criterion C (small popn size and decline)	Criterion D (very small or restricted popn)	Criterion E (quantitative analysis)	OFFICIAL IUCN CATEGORY AND CRITERION
North West Atlantic	Least Concern	Not assessed	Least Concern	Least Concern	Least Concern	Not assessed	Least Concern
North East Atlantic	Data Deficient	Not assessed	Endangered B2ab(iii)	Least Concern	Vulnerable D2	Not assessed	Endangered B2ab(iii)
Mediterranean	Least Concern	Not assessed	Least Concern	Least Concern	Least Concern	Not assessed	Least Concern
South West Atlantic	Least Concern	Not assessed	Least Concern	Least Concern	Least Concern	Not assessed	Least Concern
North West Indian	Endangered A2b	Critically Endangered A4b	Least Concern	Least Concern	Least Concern	Not assessed	Critically Endangered A4b
North East Indian	Data Deficient	Not assessed	Endangered B2ab(iii)	Data Deficient	Critically Endangered D	Not assessed	Critically Endangered D
South West Indian	Least Concern	Not assessed	Near Threatened B2	Least Concern	Least Concern	Not assessed	Near Threatened B2
South East Indian	Data Deficient	Not assessed	Near Threatened B2	Data Deficient	Least Concern	Not assessed	Near Threatened B2
North Pacific	Least Concern	Not assessed	Least Concern	Least Concern	Least Concern	Not assessed	Least Concern
South Pacific	Critically Endangered A2b	Not assessed	Least Concern	Not assessed	Least Concern	Not assessed	Critically Endangered A2b
GLOBAL	Vulnerable A2b	Not assessed	Least Concern	Least Concern	Least Concern	Not assessed	Vulnerable A2b

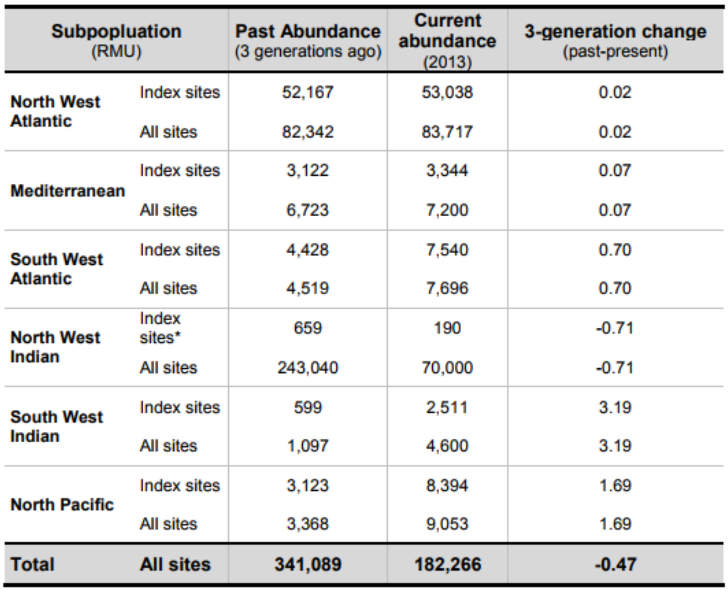
At a global level, the IUCN Red List process has assessed the loggerhead turtle as Vulnerable (Casale, 2015, Casale & Tucker, 2017) due to an estimated decline of 47% over the last three generations (Tables 2.2.1 & 2.2.2). This can mostly be attributed to the severe decline in the North West Indian subpopulation, as all other subpopulations show some increase. Analysis of more recent trends confirm that most of the subpopulations are showing a positive trend (Mazaris *et al.,* 2017). Factors likely to have contributed to recent growth in abundance at many sea turtle nesting sites are reduction of illegal harvesting of eggs and protection from predators. Reduced harvesting of turtles at sea, methods to reduce bycatch in fishing gear and international agreements prohibiting trade in sea turtle productions have also benefited this species (Mazaris *et al.,* 2017).

Step 1.4 - Analysis of distribution and trends

For marine turtles, annual counts of nesting females and their nesting activities (more often the latter) are the most frequently recorded and reported abundance metric across index monitoring sites, species, and geographic regions. There are some recognized limitations with this approach, for example because nesting females are a very small proportion of a sea turtle population, and uncertainty about many of the demographic parameters such as survivorship across life stages and generation length. Monitoring effort and methodologies can also vary within and across study sties complicating comparisons. A total of about 200,000 clutches are laid annually with an estimated 36,000- 67,000 nesting females annually (Casale & Tucker, 2017).

Six sub-populations out of the ten described (North West Atlantic, Mediterranean, South West Atlantic, North West Indian, South West Indian, North Pacific) comprise about 90% of the current annual nests globally. Trends estimated on time series datasets with more than 10 years of data on nesting activities at 153 index nesting sites reveal different trends with an estimated decline globally of 47% over the past three generations (Table 2.2.2).

***Table 2.2 2** Overall trends of the six Loggerhead Turtle subpopulations for which past and present abundance values are available (such data are not available for the other four subpopulations including the Nort East Atlantic)(from Casale & Tucker, 2017)*



Subpopluation (RMU)		Past Abundance (3 generations ago)	Current abundance (2013)	3-generation change (past-present)
North West Atlantic	Index sites	52,167	53,038	0.02
	All sites	82,342	83,717	0.02
Mediterranean	Index sites	3,122	3,344	0.07
	All sites	6,723	7,200	0.07
South West Atlantic	Index sites	4,428	7,540	0.70
	All sites	4,519	7,696	0.70
North West Indian	Index sites*	659	190	-0.71
	All sites	243,040	70,000	-0.71
South West Indian	Index sites	599	2,511	3.19
	All sites	1,097	4,600	3.19
North Pacific	Index sites	3,123	8,394	1.69
	All sites	3,368	9,053	1.69
Total	**All sites**	**341,089**	**182,266**	**-0.47**

Conclusions

The loggerhead turtle is exposed to different threats at different stages of its life cycle. At sea, by-catch in fisheries, boat strikes and pollution are the major threats. Onshore, nesting females are disturbed by traffic, beachfront lighting and noise, uncontrolled development and construction. As a consequence, there has been an overall decline in the population of this species at a global level, at least over the last three generations, and it has been classified as Vulnerable on the IUCN Red List.

The total population size of loggerhead turtle is unknown but it has been estimated that there are between 36,000-67,000 nesting females annually. Trends vary across the sub-populations but the current estimate is that over the last three generations there has been a reduction of around 47%. This can mostly be attributed to the severe decline in the North West Indian subpopulation, as all other subpopulations show some increase. Analysis of more recent trends confirm that most of the subpopulations are currently showing a positive trend.

Step 2.1 - FRP assessment (Mediterranean)

Loggerhead turtles are believed to have colonized the Mediterranean from stocks in the western Atlantic, and the rookeries believed to have become isolated from the Atlantic populations at the beginning of the Holocene (Encalada *et al.,* 1998). Ten sub-populations have been identified globally including a Mediterranean subpopulation although adult loggerhead turtles from the North East Atlantic do occur in the western Mediterranean. Several genetic nesting stocks have been identified within the Mediterranean sub-population (Carreras *et al.* 2007, Clusa *et al.* 2013, Garofalo *et al.* 2013), but metapopulation dynamics support its designation as a single sub-population, or regional management unit (Wallace *et al.* 2010).

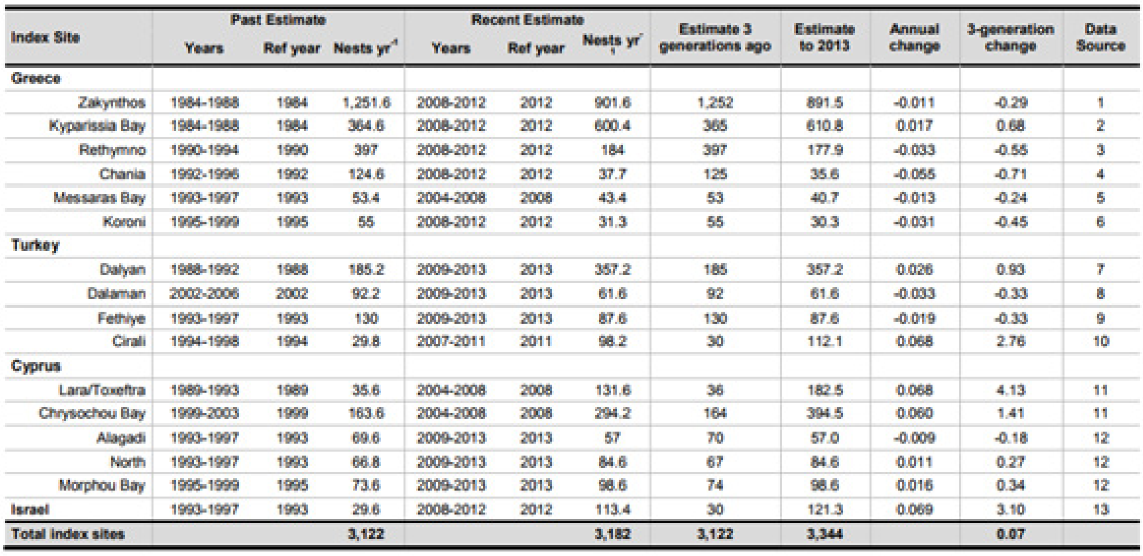
In the Mediterranean, the main threats are considered to be fishery bycatch and the degradation of nesting habitat due to coastal development (Casale & Margaritoulis, 2010). In the eastern Mediterranean, there was severe exploitation of turtles until the mid-1960s. Sella (1982) estimated at least 30,000 turtles (green and loggerheads) were caught offshore of what is now Israel from the end of the First World War until the mid-1930s for example. Loggerhead turtles were also taken in other locations such as at Iskendeurn Bay and Isole Eolie north of Sicily where an estimated 500-600 turtles were caught annually. Fishing of turtles has since been prohibited in many Mediterranean countries but even where prohibited they may be trade. For example, Laurent *et al.*, (1996) estimated that several thousand turtles (around 68% loggerheads) are probably killed annually in Egypt.

There are no historical data on population trends but the Mediterranean subpopulation is known to have been affected and declined due to incidental catch in fisheries, egg harvesting and tourist development on and around nesting beaches (Margaritoulis *et al.,* 2003). There are certainly anecdotal reports of a decrease in numbers compared to previous decades particularly at nesting sites, for example on beaches around Croatia, Italy, Sicily, Greece and the Mediterranean coast of Egypt (Casale *et al.,* 2008). There are also anecdotal reports from fishermen of declines in the abundance of sea turtle in marine areas, for example around Sicily (Casale *et al.,* 2007).

More recent trends, based on anecdotal and survey data, suggest decreasing, stable and increasing trends in different areas. Nest counts, mostly from protected beaches show an overall increase over the past three generations and IUCN has assessed the status of the loggerhead turtle subpopulation in the Mediterranean as being of 'Least Concern'.

In 2013, the number of nests in the Mediterranean subpopulation was estimated to exceed 7,200 per year (Casale & Margaritoulis, 2010). Data on nesting activity over the last ten years at 16 sites considered to be representative, show different trends, but when considered together for the entire Mediterranean, they show an overall increase over the past three generations of 7% per year (Table 2.2.3, Casale, 2015). The assessment should however be treated with caution as the 16 sites host less than half of the annual nests of the Mediterranean subpopulation. Updated analysis using only time series data on abundance and population trends, where the most recent published annual abundance was from 2010 or later, also reveals an upward trend in loggerhead turtle in the Mediterranean sub-population (Mazaris *et al.,* 2017).

Table 2.2.3 *Summary of information on past and present nesting abundances and trends at index nesting sites of the Mediterranean subpopulation. Values are average numbers of nests per year. Past estimates were assumed to reflect abundance three generations prior, and thus used in calculation of annual and 3-generation declines (from Casale, 2015)*



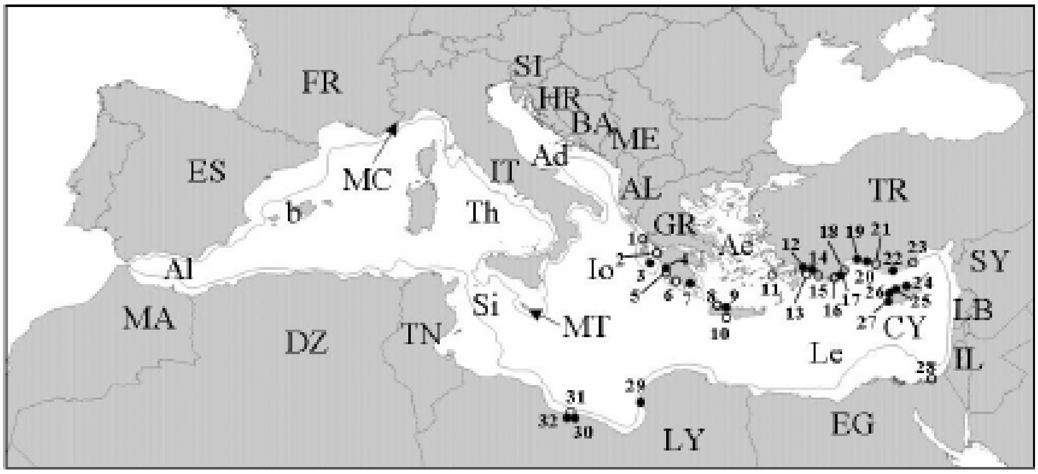
Index Site	Past Estimate			Recent Estimate			Estimate 3 generations ago	Estimate to 2013	Annual change	3-generation change	Data Source
	Years	Ref year	Nests yr^{-1}	Years	Ref year	Nests yr^{-1}					
Greece											
Zakynthos	1984-1988	1984	1,251.6	2008-2012	2012	901.6	1,252	891.5	-0.011	-0.29	1
Kyparissia Bay	1984-1988	1984	364.6	2008-2012	2012	600.4	365	610.8	0.017	0.68	2
Rethymno	1990-1994	1990	397	2008-2012	2012	184	397	177.9	-0.033	-0.55	3
Chania	1992-1996	1992	124.6	2008-2012	2012	37.7	125	35.6	-0.055	-0.71	4
Messaras Bay	1993-1997	1993	53.4	2004-2008	2008	43.4	53	40.7	-0.013	-0.24	5
Koroni	1995-1999	1995	55	2008-2012	2012	31.3	55	30.3	-0.031	-0.45	6
Turkey											
Dalyan	1988-1992	1988	185.2	2009-2013	2013	357.2	185	357.2	0.026	0.93	7
Dalaman	2002-2006	2002	92.2	2009-2013	2013	61.6	92	61.6	-0.033	-0.33	8
Fethiye	1993-1997	1993	130	2009-2013	2013	87.6	130	87.6	-0.019	-0.33	9
Cirali	1994-1998	1994	29.8	2007-2011	2011	98.2	30	112.1	0.068	2.76	10
Cyprus											
Lara/Toxeftra	1989-1993	1989	35.6	2004-2008	2008	131.6	36	182.5	0.068	4.13	11
Chrysochou Bay	1999-2003	1999	163.6	2004-2008	2008	294.2	164	394.5	0.060	1.41	11
Alagadi	1993-1997	1993	69.6	2009-2013	2013	57	70	57.0	-0.009	-0.18	12
North	1993-1997	1993	66.8	2009-2013	2013	84.6	67	84.6	0.011	0.27	12
Morphou Bay	1995-1999	1995	73.6	2009-2013	2013	98.6	74	98.6	0.016	0.34	12
Israel	1993-1997	1993	29.6	2008-2012	2012	113.4	30	121.3	0.069	3.10	13
Total index sites			**3,122**			**3,182**	**3,122**	**3,344**		**0.07**	

Conclusions

There are no historical data on population trends of the Mediterranean subpopulation of loggerhead turtle, but anecdotal reports suggest a decrease in numbers compared to previous decades particularly at nesting sites. In 2013, the number of nests in the Mediterranean sub-population was estimated to exceed 7,200 per year. Data on nesting activity over the last ten years at 16 sites considered to be representative, show different trends but when considered together for the entire Mediterranean, they show an overall increase of 7% per year compared to the situation three generations ago.

As the Mediterranean subpopulation is believed to be recovering from a period of exploitation and impact from other human activities FRP > CV.

Step 2.2 - FRR assessment (Mediterranean)
The Mediterranean subpopulation of the loggerhead turtle ranges throughout this regional sea with the highest density in the western Mediterranean (from the Alboran Sea to the Balearic Islands), the Sicily Strait, the Ionian Sea and the continental shelves in the north Adriatic, off Tunisia-Libya, Egypt, and off southeast coast of Turkey. Individuals from the North East Atlantic subpopulation are also known to occur in the western Mediterranean, intermingling with adults from the Mediterranean subpopulation (Casale *et al.,* 2008).



Major nesting sites (≥50 nests/yr) of *Caretta caretta* in the Mediterranean (based on country chapters): 1, Lefkas Isl; 2, Kotychi; 3, Zakynthos Isl.; 4, Kyparissia Bay; 5, Beaches adjacent to Kyparissia town; 6, Koroni; 7, Lakonikos Bay; 8, Bay of Chania; 9, Rethymno; 10, Bay of Messara; 11, Kos Isl.; 12, Dalyan; 13, Dalaman; 14, Fethiye; 15, Patara; 16, Kale; 17, Finike-Kumluca; 18, Çıralı; 19, Belek; 20, Kızılot; 21, Demirtaş; 22, Anamur; 23, Göksu Delta; 24, Alagadi; 25, Morphou Bay; 26, Chrysochou Bay; 27, Lara/Toxeftra; 28, Areash; 29, Al-Mteafla; 30, Al- Ghbeba; 31, Al-thalateen; 32, Al-Arbaeen. Closed circles: >100 nests/yr; open circles: 50-100 nests/yr. The 200-m bathymetry line is shown. Country codes according to the International Organization for Standardization (ISO). AL: Albania; DZ: Algeria; BA: Bosnia and Herzegovina; HR: Croatia; CY: Cyprus; EG: Egypt; FR: France; GR: Greece; IL: Israel; IT: Italy; LB: Lebanon; LY: Libya; MT: Malta; MC: Monaco; ME: Montenegro; MA: Morocco; SI: Slovenia; ES: Spain; SY: Syria; TN: Tunisia; TR: Turkey. Ad: Adriatic; Ae: Aegean; Al: Alboran Sea; Io: Ionian; Le: Levantine basin; Si: Sicily strait; Th: Thyrrenian; b: Balearic.

Figure 2.2.5 *Major nesting sites of loggerhead turtle in the Mediterranean (from Casale & Margaritoulis, 2010)*

Adults from the Mediterranean subpopulation mostly nest in the eastern Mediterranean basin predominantly in Greece, Turkey, Cyprus and Libya (Figure 2.2.5). There has been no recorded nesting in the eastern Adriatic although occasional nesting cannot be ruled out especially in the past (Casale & Margaritoulis, 2010). There is a lack of historical data on the past distribution and abundance of nesting beaches but anecdotal reports of more nesting activity in the recent past, for example on beaches around Croatia, Italy, Greece and the Mediterranean coast of Egypt (Casale *et al.,* 2008).

Conclusions

The loggerhead turtle has a widespread distribution in the Mediterranean although nesting beaches are largely confined to the eastern part. The numbers of females using particular nesting beaches has fluctuated and although there are anecdotal reports of more widespread nesting this has cannot be quantified. No reduction in range of foraging adults has been suggested, even when the population was less abundant. The current range encompasses the entire Mediterranean basin.

It can therefore be concluded that the FRR for adults of this subspecies of the loggerhead turtle is the entire Mediterranean basin, and that FRR = CV. For nesting females there is anecdotal information to suggest that they ranged more widely therefore FRR > CV.

References

Avens, L., and M. L. Snover. 2013. Age and age estimation in sea turtles. Pages 97–134 in J. Wyneken, K. J. Lohmann, and J. A. Musick, editors. The biology of sea turtles. CRC Press, Boca Raton, Florida, USA

Bolten, A.B. (2003) Variation in Sea Turtle Life History Patterns: Neritic vs. Oceanic Developmental Stages. In Lutz, P.L., Musick, J.A. & Wyneken, J. (Eds) The Biology of Sea Turtles. Volume 2. Chapter 9. Pages 243 – 258. CRC Press, New York.

Bolten, A.B. & Witherington, B.E. 2003. Loggerhead Sea Turtles. Smithsonian Books, Washington, D.C., USA.

Bolten, A.B., Bjorndal K.A., Martins, H.R., *et al.* (1998) Trans-Atlantic developmental migrations of loggerhead sea turtles demonstrated by mtDNA sequence analyses. Ecological Applications, 8, 1–7.

Bowen, B.A., Bass, A., Soares, L., & Toonen, R.J. 2005. Conservation implications of complex population structure: lessons from the loggerhead turtle (Caretta caretta). Molecular Ecol. 14:2389-2402.

Careras, C. *et al.*, 2007 The genetic structure of the loggerhead sea turtle (Caretta caretta) in the Mediterranean as revealed by nuclear and mitochondrial DNA and its conservation implications. Conserv.Genet. DOI 10.1007/s10592-006-9224-8. http://www.archelon.gr/files/bibliography/%5B91%5DCarreras06_GeneticStructure.pdf

Casale, P. *et al.*, 2007. Incidental catch of marine turtles by Italian trawlers and longlines in the central Mediterranean. Aquatic.Cons: Mar.Fresh.Eco. 17:686-701.

Casale, P. et al 2008.Foraging ecology of loggerhead sea turtles Caretta caretta in the central Mediterranean Sea: evidence for a relaxed life hisotyr model. Mar.Ecol.Prog.Ser 372:265-276.

Casale, P. 2015. Caretta caretta (Mediterranean subpopulation). The IUCN Red List of Threatened Species 2015: e.T83644804A83646294. http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T83644804A83646294.en. Downloaded on 18 September 2017.

Casale, P. & Margaritoulis, D.(Eds) 2010. Sea turtles in the Mediterranean: Distribution, threats and conservation priorities. Gland Switzerland, IUCN. 294pp.

Casale, P. & Tucker, A.D. 2017. Caretta caretta. (amended version published in 2015) The IUCN Red List of Threatened Species 2017: e.T3897A119333622. Downloaded on 18 September 2017.

Clusa, M., Carreras, C., Pascual, M., *et al.*, 2013. Mitochondrial DNA reveals Pleistocenic colonisation of the Mediterranean by loggerhead turtles (Caretta caretta). J. Exp.Mar. Biol.Ecol.439: 15-24.

Encalada. S.E. *et al.* 1998. Population structure of loggerhead turtle (Caretta caretta) nesting colonies in the Atlantic and Mediterranean as inferred from mitochondrial DNA control region sequences. Marine Biol. 130:567-575.

Garofalo, L., Mastrogiacomo, A., Casale, P., *et al.*, 2013. Genetic characterization of central Mediterranean stocks of the loggerhead turtle (Caretta caretta) using mitochondrial and nuclear markers, and conservation implications. Aquatic Cons: Marine and Freshwater Ecosystems 23: 868-884.

Laurent, L., Abd El-Mawla, E.M, Bradai, M.N. *et al.*, (1996) Reducing sea turtle mortality induced by Mediterranean fisheries: trawling activity in Egypt, Tunisia and Turkey, Report for the WWF Mediterranean Programme. WWF Project 9E0103. 32pp.

Laurent, L., Casale, P., Bradai, M.N. *et al.* 1998 Molecular resolution of the marine turtle stock composition in fishery bycatch: a case study in the Mediterranean. Molecular Ecol. 7:1529-1542.

Margaritoulis, D. *et al.*,2003. Loggerhead turtles in the Mediterranean: present knowledge and conservation perspectives. Chapter 11 in Bolten, A.B. & Witherington (Eds). Loggerhead Sea Turtles. Smithsonian Institution Washington. page 175-198.

Mazaris, A.D. *et al.*, 2017. Global sea turtle conservation successes. Science Advances:3: e1600730 20 September 2017.

McClellan, C.M. & Read, A.J. 2007. Complexity and variation in loggerhead sea turtle life history. Biol.Lett. Dec 22:3(6): 592-594.

Sella, I. 1982 Sea turtles in the eastern Mediterranean and northern Red Sea. In K. A. Bjorndal (ed.). Biology and conservation of sea turtles, 417–423. Washington, D.C.: Smithsonian Institution Press.

Wallace, B.P. *et al.*, 2010. Regional Management Units for Marine Turtles: A Novel Framework for Prioritizing Conservation and Research across Multiple Scales. PLoS ONE Vol 5(12). http://journals.plos.org/plosone/article?id=10.1371/journal.pone.0015465

3 Terrestrial mammals

Michela Pacifici, Carlo Rondinini & Luigi Boitani

3.1 Lesser horshoe bat (*Rhinolophus hipposideros*) in Italy

Step 1.1 - Biology of the species

Lesser horseshoe bats (*Rhinolophus hipposideros*) are the smallest of the European horseshoe bats, weighting only 5-9 grams and with a wingspan of 192 to 254 mm. The mating system of the lesser horseshoe bat is not well known. They breed in the autumn, producing 1 or 2 offspring. Average lifespan in the wild is 3-4 years. The lesser horseshoe bat has a wide range in the western and central Palaearctic. It is found in almost all the European countries, North Africa and the Middle east (Taylor 2016). In Italy, the species is present throughout the territory including some small islands (Capraia, Elba, Giglio, San Pietro, Pantelleria)(Figure 3.1.1).

Figure 3.1.1 *Distribution of the lesser horshoe bat in Itay (modified from Stoch & Genovesi 2016)*

In Italy in 2003 there were 99 temporary roosting sites (max n. of individuals per site = 20), 88 hibernacula (max n. of individuals per site = 3) and 30 nurseries (max n. of individuals per site = 150; Gruppo Italiano Richerca Chirotteri 2005). Individuals are usually solitary, except when forming breeding roosts in the summer. Reproductive colonies number 2-100 females and a few males (Lanza 2012). During winter hibernation, these bats prefer to roost individually, but in proximity to others (Schober and Grimmberger 1997).

Step 1.2 - Spatial scale of functioning

The species is sedentary. Winter and summer refugia are located within 5-20 km distance from each other, but sometimes this distance can be much larger (Lanza 2012). The maximum distance covered by an individual (282 km) was recorded by Arthur and Lemaire (2009) with no further details.

Conclusions

This species could be listed under population category S1 as this bat is sedentary, widespread and living in discrete colonies/subpopulations with a certain exchange resulting in one effective population at the national (Italian) level.

Step 1.3 - Historical perspective: what happened to the species?

Since the 1950s, the northern border of the species' range in central and western Europe has moved southwards. The species became extinct in the Netherlands, northern Belgium, most of Germany and Gibraltar. In Poland, 87% of the hibernating population became extinct between 1950 and 1990. In Austria and Switzerland colonies remained only at higher elevations (>400 m), although in Switzerland the species has started to recover over the last 10 years (from 2200 to 2500 of adults contained in maternity roosts). Also in the Czech Republic the species is recovering, which might reflect a general trend in central Europe (Taylor 2016). The mechanisms responsible for this decline have not been conclusively identified, but since some populations of *Pipistrellus pipistrellus* have increased substantially, possibly as a consequence of foraging upon insects attracted by street lamps, one of the causes might be the increased competition with *P. pipistrellus* (Arlettaz *et al.* 2000). Other possible causes are the loss of foraging habitat and the increasing use of pesticides in central and northern Europe (Bontadina *et al.* 2002).

Step 1.4 - Analysis of distribution and trends

For Italy, Lanza in 1959 described the species as widespread and frequent, but the data currently available for some regions (e.g. Tuscany and northern Piemonte) indicate that the species has undergone a decline in the geographic range in the last decades. The loss of foraging sites due to an increase in agricultural practices and in the use of pesticides has been one of the main causes of the reduction in population size for the species. Like congeneric species, it is very likely that *R. hipposideros* suffers from habitat loss due to deforestation of the flat areas in northern Italy. Populations in caves and other underground habitats have suffered from increased disturbance (for example by tourist visits to caves). In addition, the species suffered from the loss of summer refugia in buildings (Rondinini *et al.* 2013). Significant reductions in the distribution of the species have been reported for Tuscany in the last 60 years, Campania and Aosta Valley regions. In Piemonte, the species was known to be widespread and common but current observations are sporadic and related to single individuals, limited to the pre-alpine (Biella district) and Apennine reliefs (Alessandria district; Toffoli 2005). Despite this general trend of decline, the species is increasing its populations in Liguria and southern Piemonte (Lanza 2012). In fact, while in the 1970s the *R. hipposideros* was present in only 4 caves, in 2003 and 2005 it has been recorded in almost all the Guardiabella caves (Imperia). In these areas, the species has become recently more abundant to the detriment of *R. ferrumequinum* with respect to the historical situation (Lanza 2012). In February 2000 150 individuals were counted only in one cave (Sgarbu du Ventu).

Given its proved ability to recover in some of the areas in central Europe, Liguria and southern Piemonte there might be scope for restoration.

Despite estimates of the total population size are unavailable for the species in Italy, according to Rondinini *et al.* (2013), the lesser horseshoe bat has undergone a decline in its population size of almost 50% in the last 3 generations (30 years). Several colonies disappeared in the last years, especially in Campania (2 colonies out of 5 in a well-managed protected area since 1998) and the only known reproductive colony in Aosta Valley. It has experienced a significant reduction in geographic range size in Tuscany since the 1950s (the remaining range is probably <1% with respect to the original one). Despite the overall decline in population size and distributional range, in the last 15 years the species has shown increases in population and range size in at least some sites in Liguria and Piemonte.

Conclusions

The lesser horseshoe bat is known to have been much more widespread and abundant in Europe in previous decades and in the recent past. Substantial declines occurred starting from the second half of the 20th century. This was principally due to the loss of foraging sites, habitat conversion and the use of pesticides. Other causes like human disturbance and deforestation have played a role in population reduction in most of the regions in Italy, with information of steep declines and range contractions for Tuscany and most of Piemonte regions. The magnitude of the overall decline in Italy is almost 50% in the last 30 years, with some exceptions of recovery where the species has substituted R. ferrumequinum in caves (> 150 individuals counted in only one cave near Imperia in 2000). In Tuscany, the current population size is <1% with respect to the population in the 1950s.

Step 2.1 - FRP assessment

In general, in Italy, the present day population of the lesser horseshoe bat is known to be substantially reduced from that of previous decades and significant distribution have been lost in the regions Tuscany, Campania, Piemonte and Aosta Valley. Based on the information available on the major reproductive colonies in Italy (Lanza 2012), and the areas in which the species is either lost or known to be expanding, at least three long-term viable populations (with > 550 individuals) must be restored, located in northern (Piemonte or Liguria), central (Tuscany or Lazio), and southern (southern Campania) Italy, in order to ensure a long-term viable population in Italy covering the geographical variation within its natural range. Despite the species is spread across the country, its population has extremely reduced and not viable in many parts of the range. The proposed extent of the three viable populations (around 40,000 km^2) should ensure the current ecological variation within the species range, since these areas were historically the most densely populated by lesser horseshoe bat individuals.

Population estimates for the species in Italy are unknown, but Article 17 reporting suggests that the FRP in the 3 biogeographic regions in the country (Alpine, Mediterranean and Continental) must be larger than current value. Since no estimates exist, we used the method in Hilbers *et al.* (2016) to estimate a mean MVP target of 547 individuals, given an estimated body mass value of 0.00471 kg. We then used considerations about decline rate and loss of geographic (historical) variations within the natural range for upscaling to the national (species) level, more specifically 1) requiring a population representing past known geographical coverage in Italy (see above), 2) requiring a population representing local climatic adaptation and genetic variation, for example, several populations of small mammals in southern Italy are genetically distinct from the ones in central-North Italy, and are likely to be separate species, and 3) considering feasibility (pressures and remaining natural habitat).

Conclusions

Although estimates of population size for Italy are almost unknown, there is evidence that both in the long and in the short term, the species was much more common and with a much larger distribution in the past than when the Directive came into force. Since the population of the species extends in the 3 biogeographic regions of Italy (Alpine, Mediterranean and Continental), which are ecologically different in terms of habitats for the species, we suggest to have at least one viable population in each biogeographic region to ensure the current ecological variation within the species range, following Hilbers *et al.*'s method:
FRP >> CV (suggested value about 550*3 = 1650 individuals).

Step 2.2 - FRR assessment

We know from the literature that the past range of the species was much larger than the 1994-value, occupying almost the whole area of the country. From Figure 3.3.1 is instead evident that the species is currently concentrated in central Italy, with spotted populations in the northern and in the southern portions of the country, we set FRR >> DV, i.e. Italy extent. According to the Art. 17 reporting in 2013, the gridded distributional area for the species in Italy corresponds to 213,100 km^2 (~2/3 of the

Italian territory), and the current range size in the 3 biogeographic regions is considered as favourable.

Conclusions

Even if the species is currently present in 2/3 of the Italian territory, and FRR might be current value, we know that the species occupied the whole country since the 1950s. Therefore we suggest that :
213,000 km^2 (CV) ≤ FRR ≤ 301,000 km^2 (historic range).

References

Arthur, L. & Lemaire, M. (2009). Les Chauves-souris de France Belgique Luxembourg et Suisse. Biotope.

Arlettaz, R., Godat, S., and Meyer, H. (2000). Competition for food by expanding pipistrelle bat populations (Pipistrellus pipistrellus) might contribute to the decline of lesser horseshoe bats (Rhinolophus hipposideros). Biological Conservation, 93(1), 55-60.

Gruppo Italiano Ricerca Chirotteri (GIRC), 2005. The Italian Bat Roost Project: a preliminary inventory of sites and conservation perspectives. Hystrix, (n.s.), 15(2)[2004]:55-68.

Hilbers, J. P., Santini, L., Visconti, P., Schipper, A. M., Pinto, C., Rondinini, C., Huijbregts, M. A. (2016). Setting population targets for mammals using body mass as a predictor of population persistence. Conservation Biology. DOI: 10.1111/cobi.12846

Lanza B. (2012): Mammalia V, Chiroptera, Fauna d’Italia, Vol. 46, Edizioni Calderini de «Il Sole 24 ORE», Milano, XIII + 786 pp

Rondinini, C., Battistoni, A., Peronace, V., Teofili, C. (2013). Lista rossa IUCN dei vertebrati italiani. Comitato Italiano IUCN e Ministero dell’Ambiente e della Tutela del Territorio e del Mare, Roma, 56.

Schober, W., E. Grimmberger. (1997). The Bats of Europe and North America. New Jersey: TFH Publications, Inc.

Stoch F., Genovesi P. (ed.), 2016. Manuali per il monitoraggio di specie e habitat di interesse comunitario (Direttiva 92/43/CEE) in Italia: specie animali. ISPRA, Serie Manuali e linee guida, 141/2016.

Taylor, P. 2016. Rhinolophus hipposideros. The IUCN Red List of Threatened Species 2016: e.T19518A21972794. http://dx.doi.org/10.2305/IUCN.UK.2016-2.RLTS.T19518A21972794.en. Downloaded on 13 January 2017.

Toffoli, R. (2005). Il Rhinolophus hipposideros in provincia di Cuneo: status e distribuzione. Memorie del Museo della Riserva Naturale Orientata di Onferno 2 (2005).

3.2 Cabrera's vole (*Microtus cabrerae*) in the Iberian peninsula

Step 1.1 - Biology of the species
Microtus cabrerae is a medium-sized rodent belonging to the Arvicolinae subfamily (with voles, lemmings and muskrats). Among the members of the genus it has one of the largest body masses (mean weight is 52 g, ranging between 30 and 78 g; Ventura *et al.* 1998). The coat has a rough, brownish olive colour on the dorsal side and yellowish tinge on the ventral side.

Although *Microtus cabrerae* may reproduce throughout the year (Pita *et al.* 2010), breeding activity is higher during the winter months, and in very dry years reproduction may even cease completely at the end of the spring and the beginning of summer (Ventura *et al.* 1998; Rosário 2012). This reduction or cessation of reproduction during the months with lowest precipitation and highest temperatures has been associated with major declines in food quantity and quality (Ventura *et al.* 1998; Fernández-Salvador *et al.* 2005). Mean number of embryos varies from 3 to 7, which tends to decline from winter to summer (Ventura *et al.* 1998), but mean litter size observed in captive animals ranges from 3 to 5 individuals, thus suggesting a possible prenatal mortality (Fernández-Salvador *et al.* 2001). Gestation length is about 23–24 days, and sex ratio of captive neonates does not differ from 1:1 (Fernández-Salvador *et al.* 2001).

Step 1.2 - Spatial scale of functioning
The species is endemic to the Iberian peninsula (Portugal and Spain; Fernandes *et al.* 2008; Figure 3.2.1). The small mammal has low mobility and occurs in vast areas, fragmented due to the intensification of agriculture in the Iberian Peninsula since the Iron Age.

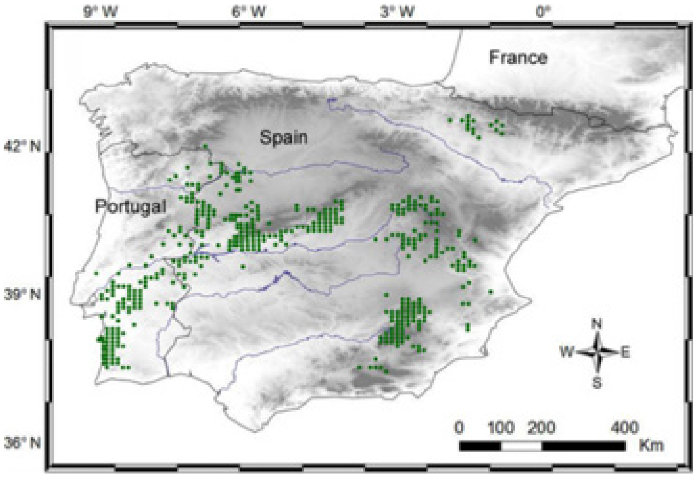


Figure 3.2.1 *Current distribution of Microtus cabrerae. Green dots represent occurrence points. From: Microtus cabrerae (Rodentia: Cricetidae, Mamm Species. 2014;46(912):48-70. doi:10.1644/912.1)*

Conclusions

Population category is S6: a small, resident species with low mobility and scattered (often fragmented) distribution. For this category FRVs will be set at the national level.

Step 1.3 - Historical perspective: what happened to the species?
Microtus cabrerae is an Iberoccitane endemic species, originally occupying the Iberian peninsula and southern France (Ayarzagüena & López-Martínez 1976; Cabrera-Millet *et al.* 1983; Sesé *et al.* 2011). The geographical area of the species has reduced from the late Pleistocene (about 0.13–0.01 million years ago) to recent times, and its current distribution range is restricted to the Iberian Peninsula

(Ayarzagüena & López-Martínez 1976; Cabrera-Millet *et al.* 1982; Fernandes *et al.* 2008; Figure 1). The first range contraction of *Microtus cabrerae* was observed during the Last Glacial Maximum (late Pleistocene, about 24 thousand years ago), and was followed by an expansion phase during the Holocene Climatic Optimum (c. 8,000 years ago; Laplana & Sevilla 2013). A subsequent range contraction took place during the Bronze Age, probably due to climatic warming and increase in aridity (5,000–2,500 years ago; Cabrera-Millet *et al.* 1982; Fernández-Salvador 1998). However, population retraction may have been particularly intense from the Iron Age (c. 2,000 years ago) to present, suggesting that agricultural expansion has been the main cause of the reduction and fragmentation of the distribution of *M. cabrerae* since that period (Garrido-García & Soriguer-Escofet 2012; Laplana & Sevilla 2013), which has resulted in the destruction of suitable habitat for the species.

Step 1.4 - Analysis of distribution and trends

The conservation status of *Microtus cabrerae* reported by the International Union for Conservation of Nature changed from "Lower Risk/Near Threatened" in 1996 to "Near Threatened" in 2008, based on its restricted geographic range and area of occupancy (c. 2,000 km^2), resulting from the severe fragmentation of suitable habitat and subsequent population declines across many areas where the species was present in the past (Fernandes *et al.* 2008). Despite the species has been recently found in some new localities where it was not previously known (e.g., Garrido-García *et al.* 2008; Ortuño 2009), empirical observations suggest an overall declining trend of the species at both the regional and local scales (Fernández-Salvador 2007). For instance, in Spain it is estimated that over the past 10 years, about one-third of populations of *M. cabrerae* from Cuenca, Toledo, Albacete, and Madrid have disappeared or strongly reduced. Likewise, populations from Andalucia, Huesca, Zaragoza, Navarra, and Zamora also have undergone strong reductions and high rates of local extinction (Fernández-Salvador 2007). In Portugal, population trends are unknown, but recent steep declines are suspected across most of the distribution of the species within the country (Queiroz *et al.* 2005). Over the past years large amounts of suitable habitat across the species' distribution range have been replaced by intensively managed fields mostly devoted to crop and livestock production (Pita *et al.*2006). In addition, the particularly severe and exceptional droughts affecting the region (e.g., in 2005 and 2012), resulted in considerable reductions in populations of *the species*, and extremely low recovery abilities (Pita *et al.* 2014). At the national level the species is thus currently classified as "Vulnerable" in both Spain and Portugal.

Population densities are moderate by comparison with other arvicoline rodents, typically varying between 17 and 350 individuals per hectare (Palomo and Gisbert 2002). Estimates of the total population size are unavailable for the species. ES reports 1000-5000 colonies for the period 2007-2012. The observed declines in population size in several areas in Spain exceed 1/3 of the population recorded 20 years ago. This is particularly true in the regions in southern Spain. Agricultural intensification, including overgrazing, has probably contributed to range reductions and fragmentation over the last few decades (Palomo 1999).

Conclusions

The Cabrera's vole is known to have been much more widespread and abundant in the Iberian peninsula in the recent past. Substantial declines occurred starting from the historical periods due to the intensification of agriculture and overgrazing, and the subsequent fragmentation of suitable habitats for the species. Other causes of decline may be the increased pressure on streams and other wetland areas where the species occurs. There is suspicion that interspecific competition with Arvicola sapidus may be a problem (Pita *et al.*2006). The species may also be severely, negatively affected by global warming.

The historical distribution and abundance of this species was significantly greater in the past than that of present day. The magnitude of the overall decline, especially in Spain, is almost 33% in the last 20 years, with some exceptions of colonisations or new records.

Step 2.1 - FRP assessment

Populations are grouped as colonies, often restricted to areas of reduced dimensions (often <500 m^2; Santos *et al.* 2006). Population density has been recorded in some areas as 17-115 ind/ha

(Fernández-Salvador & Ventura 2005). Therefore, 1 ha=20 colonies (~0.85-5.75 individuals per colony).

During the 2012 and 2014, the project "Bases para la conservación del Iberón, o Topillo de Cabrera (Iberomys cabrerae), en España" started visiting known colonies of the species in order to evaluate the its conservation status. The project is unfinished and they count only with partial data. Results obtained so suggest that the number of colonies extinguished reaches from 18% to 96%, compared to those existing 5-10 years ago. Spain reports that the number of colonies for a FRP should be >1000 (850-5750 individuals), and at least 26 colonies have been sampled in Portugal (Santos *et al.* 2006).

Conclusions

For ES MED: FRP>>850-5750 (CV), according to 18%-96% reductions in the last 10 years.
For PT MED: FRP>>22-150 individuals.

Step 2.2 - FRR assessment

Portugal reports that the current size of the range (30000 km^2) can be considered as favourable, while for Spain the current value of 50444 km^2 seems to be inadequate. Due to the increasing intensification of agriculture and overgrazing in most of the suitable habitats occupied by the species, it is very likely that the past range of the species was much larger than the 1994 value, we set FRR at least as the extent of the open woodlands in Spain (the areas of its occurrence in Portugal include cork oak (*Quercus suber*) and holm oak (*Q. rotundifolia*) open woodlands as the main habitat) + the current area occupied by the species in Portugal.

Conclusions

For ES MED >> CV.
For PT MED = CV.

FRR >> CV. This estimate of the FRR corresponds to the majority of its (former) natural range.

References

Ayarzagüena, J., López-Martínez, N. 1976. Estudio filogenético y comparativo de Microtus cabrerae y Microtus brecciensis. Doñana Acta Vertebrata 3:181–204.

Cabrera-Millet, M., Britton-Davidian, J., Orsini, P. 1982. Génétique biochimique comparée de Microtus cabrerae Thomas, 1906 et de trois autres espèces d'Arvicolidae mediterranéens. Mammalia 46:381–388.

Cabrera-Millet, M.N., López-Martínez, N., Michaux, J. 1983. Un exemple de lignee endemique iberoccitane, les campagnols Microtus brecciensis et Microtus cabrerae (Mammalia, Rodentia): etude phylogenetique et contexte ecologique d'un phenomene evolutif recent. Pp. 69–83 in Actes du Symposium Paleontologique (Buffetaut E. Mazin J. M. Salmon E.eds.). Impressions le Serpentaire, Montbéliard, France.

Fernandes, M., Pita, R., Mira, A. 2008. Microtus cabrerae. In International Union for Conservation of Nature. International Union for Conservation of Nature and Natural Resources Red list of threatened species. www.iucnredlist.org. Accessed 20 November 2012.

Fernández-Salvador, R., Ventura, J., & García-Perea, R. 2005. Breeding patterns and demography of a population of the Cabrera vole, Microtus cabrerae. Animal Biology, 55(2), 147-161.

Fernández-Salvador, R. 2007. Microtus cabrerae Thomas, 1906. Topillo de Cabrera. Pp. 429–433 in Atlas y Libro rojo de los mamíferos terrestres de España (Palomo J. L. Gisbert J. Blanco J. C.eds.). Dirección General para la Biodiversidad, Sociedad Española para la Conservación y Estudio de los Mamíferos - Asociación Española para la Conservación y el Estudio de los Murciélagos, Madrid, Spain.

Garrido-García, J.A., Soriguer-Escofet, R.C. 2012. Cabrera's vole Microtus cabrerae Thomas, 1906 and the subgenus Iberomys during the Quaternary: evolutionary implications and conservation. Geobios 45:437–444.

Hilbers, J. P., Santini, L., Visconti, P., Schipper, A. M., Pinto, C., Rondinini, C., & Huijbregts, M. A. 2017. Setting population targets for mammals using body mass as a predictor of population persistence. Conservation Biology, 31(2), 385-393.

Laplana C., Sevilla P. 2013. Documenting the biogeographic history of Microtus cabrerae through its fossil record. Mammal Review 43:309–322.

Palomo, L.J. 1999. Microtus cabrerae. In: A. J. Mitchell-Jones, G. Amori, W. Bogdanowicz, B. Kryštufek, P. J. H. Reijnders, F. Spitzenberger, M. Stubbe, J. B. M. Thissen, V. Vohralík and J. Zima (eds), The Atlas of European Mammals, Academic Press, London, UK.

Palomo, L.J. and Gisbert, J. 2002. Atlas de los mamíferos terrestres de España. Dirección General de Conservación de la Naturaleza. SECEM-SECEMU, Madrid, Spain.

Pita, R., Mira, A. and Beja, P. 2006. Conserving the Cabrera vole, Microtus cabrerae, in intensively used Mediterranean landscapes. Agriculture, Ecosystems and Environment115: 1–5.

Pita, R., Mira, A., Beja, P. 2014. Microtus cabrerae (Rodentia: Cricetidae), Mammalian Species, Volume 46, Issue 912, 5 December 2014, Pages 48–70, https://doi.org/10.1644/912.1

Rosário, I.T. 2012. Towards a conservation strategy for an endangered rodent, the Cabrera vole (Microtus cabrerae Thomas). Insights from ecological data. Ph.D. dissertation, University of Lisbon, Lisbon, Portugal.

Santos, S. M., Simões, M. P., da Luz Mathias, M., & Mira, A. 2006. Vegetation analysis in colonies of an endangered rodent, the Cabrera vole (Microtus cabrerae), in southern Portugal. Ecological Research, 21(2), 197-207.

Sesé, C., Panera, J., Rubio-Jara, S., Pérez-González, A. 2011. Micromamíferos del Pleistoceno Medio y Pleistoceno Superior en el Valle del Jarama: yacimientos de Valdocarros y HAT (Madrid). Estudios Geológicos 67:131–151.

Ventura, J., López-Fuster, M. J., Cabrera-Millet, M. 1998. The Cabrera vole, Microtus cabrerae, in Spain: a biological and morphometric approach. Netherlands Journal of Zoology 48:83–100.

3.3 Wolverine (*Gulo gulo*) in Scandinavia

Step 1.1 - Biology of the species
Wolverines are one of the largest members of the carnivorous mammal family Mustelidae, with body lengths of 65 to 105 cm and weights from 9 to 30 kg. Two subspecies of wolverines are known: North American wolverines (*G. gulo luscus*) and European wolverines (*G. gulo gulo*). Differences seem to be mainly genetic and probably as a consequence of the isolation of these two populations (Pasitschniak-Arts & Lariviere 1995). The European population of the wolverine is connected to the Asian Russian population along the Urals, and there are two different subpopulations: Scandinavian (Norway and Sweden) and Karelian (Finland; Arild Landa 2007).

Wolverines are generally solitary animals, with males and females coming together only briefly for mating (from May to August). Males have large home ranges, usually encompassing the home ranges of several females (Pasitschniak-Arts & Lariviere 1995). The average number of offspring is 3, with females giving birth in alternate years. Individuals reach sexual maturity at 2-3 years.

Step 1.2 - Spatial scale of functioning
The Scandinavian population is distributed mainly along the border of Norway and Sweden, with extensions into the southern Norwegian mountains, and the northern Norwegian county of Finnmark and adjacent areas of northwest Finland (the region of Lappland). There are three population segments within this range, the south Norwegian, the central population segment along the Norwegian/Swedish border, and a few animals breeding in the boreal forest areas of eastern Sweden (Arild Landa 2007; Figure 3.3.1). Males have home ranges of 600 to 1000 km^2, while females home ranges are 50 to 350 km^2.

Figure 3.3.1 *Wolverine distribution in Europe (from Large Carnivore Initiative for Europe at http://www.lcie.org/Large-carnivores/Wolverine)*

Conclusions

This species in Scandinavia (Sweden and Norway) is a good example of population category S4: resident, animal species with individuals with large home ranges (>100 km^2 up to >1000 km^2).

Step 1.3 - Historical perspective: what happened to the species?
Wolverines have a circumpolar distribution, corresponding with the Boreal zone of the northern hemisphere (Kvam *et al.* 1988). During the 19th century, wolverines disappeared from the southernmost part of their European range mainly through persecution, but also because of deforestation and other human developments. In the 1950s-1970s interval, wolverines underwent a steep decline, and it was the period of lowest abundance of the European populations. After this substantial range reduction, there is evidence of reappearance in some areas of its historical distribution (Rowland *et al.* 2003). In Scandinavia, the extent of the geographic range in this period was around 93,900 km^2 (Figure 3.3.2). Since 2012, population has expanded and the current range is c. 283,700 km^2, which means that the estimate has more than tripled in the last 40-60 years (Chapron *et al.* 2014). The current range size in Sweden is approximately 117,100 km^2, and it is considered as stable since 2001.

Figure 3.3.2 *Wolverine distribution in Europe in the 1950s-1970s. This is the historical period of lowest abundance. From Chapron et al. 2014. 1. Scandinavian population, 2 Karelian (Finnish-western Russian) population*

Step 1.4 - Analysis of distribution and trends
The Scandinavian population has shown a low genetic variability and subdivision among populations. This indicates that the Wolverine in Scandinavia has lost variation through a previous bottleneck and that the current populations are the result of a recent common genetic background (Walker *et al.* 2001, Flagstad *et al.* 2004). A recent population estimate is of 339-431 individuals in Norway (Chapron *et al.* 2014) and 670-830 in Sweden (Nilsson 2013). The population has a continuous distribution and is narrowly connected to the Finnish – western Russian population (Abramov 2016). In the 1950s-1970s, the population in Norway was around 100-150 individuals (3-fold increase), while in Sweden was around 60-100 individuals (5-fold increase; Chapron *et al.* 2014). The Scandinavian

part of the geographical European range is stable, but the population is very small according to the European Mammal Assessment in 2007; this determined for the European populations of Wolverine a category of Vulnerable, due to over-exploitation through hunting and trapping, predator-poisoning programmes and habitat resource extraction that caused the contraction of its historical range as well. The southern Norwegian population segment was naturally re-established during the late 1970s and was a result of protective legislation. This population segment has recently increased in numbers and distribution, but seems to have stabilized at around 100 individuals (Arlid Landa 2007).

From these data, we conclude that the population size of the Scandinavian population has increased 3-5 folds in the period 1950-2012 (Chapron *et al.* 2014). Density estimates vary from one individual per 200 to 500 km^2. However, in recent years, bottleneck effects have caused a reduction of genetic variability, and the general population trend is declining or stable in some portions of the species range (Arlid Landa 2007). The geographic range of the species in Scandinavia has shown a 300% increase in the last 60 years, but the species can potentially further expand and re-colonize historical territories if major threats stop.

Conclusions

The Wolverine in the Scandinavian population has experienced a general increase in both geographic range and population size in the last 60 years. In this time interval, the increase in geographic range has been c. 300%, while that of the population size varies between 300% and 500%. Recent trends indicate that the species is threatened by genetic bottlenecks and the general trend in Europe is still declining.

Step 2.1 - FRP assessment

Apparently the trend in population and range size of the Wolverine in the last 60 years has been positive, with the species re-colonizing parts of its natural range. However, the Scandinavian population is suffering due to genetic bottlenecks and small population size. Since we know that the population size of the wolverine in Scandinavia was much higher in the past, and the species has the potential to re-colonize its past range if threats stop, the FRP for the Scandinavian population of this species is > CV (current population size=919-1211 individuals). Instead of setting this number as the FRP, we use operators and suggest to constantly monitor the status of the population until range and population size are stable for a sufficient long time. If considering only the Sweden population, 600 individuals is the recommended FRP after a comprehensive study of the population between 2001-2012, therefore the conservation status of the population in the country (670-830 individuals; Nilsson 2013) should be considered as favourable.

Conclusions

For the Scandinavian population FRP > CV, while accounting only for Sweden 600 individuals can be considered as the appropriate FRP value (current population estimate 670-830), therefore FRP = CV.

Step 2.2 - FRR assessment

Based on the previous discussion on the increase in distribution and population size since the 1950s, and following the stepwise approach, we conclude that the range can still increase (i.e. likely to be higher than the current 283,700 km^2, even if FRR in Sweden is considered as favourable), if major threats to the species are controlled (in particular conflicts with humans due to depredation of domestic sheep).

Conclusions

For the Scandinavian population FRR>CV, while considering only the Sweden part of the range the FRR=CV.

References

Abramov, A.V. 2016. Gulo gulo. The IUCN Red List of Threatened Species 2016: e.T9561A45198537. http://dx.doi.org/10.2305/IUCN.UK.2016-1.RLTS.T9561A45198537.en. Downloaded on 23 August 2017.

Arild Landa (Large Carnivore Initiative for Europe). 2007. Gulo gulo. The IUCN Red List of Threatened Species 2007: e.T9561A13001123. Downloaded on 23 August 2017.

Chapron, G., Kaczensky, P., Linnell, J. D., Von Arx, M., Huber, D., Andrén, H., ... & Balčiauskas, L. 2014. Recovery of large carnivores in Europe’s modern human-dominated landscapes. science, 346(6216), 1517-1519.

Flagstad, Ø., Hedmark, E., Landa, A., Brøseth, H., Persson, J., Andersen, R., Segerström, P. and Ellegren, H. 2004. Colonization history and non-invasive monitoring of a re-established Wolverine (Gulo gulo) population. Conservation Biology 18: 676-688.

Nilsson 2013. Population viability analyses of the Scandinavian populations of bear (Ursus arctos), lynx (Lynx lynx) and wolverine (Gulo gulo), pp 1-75.

Pasitschniak-Arts, M., S. Lariviere. 1995. Gulo gulo. Mammalian Species, 499: 1-10.

Rowland, M. M., Wisdom, M. J., Wales, B. C., Johnson, D. H., Copeland, J. P. and Edelmann, F. B. 2003. Evaluation of landscape models for wolverines in the interior Northwest, United States of America. Journal of Mammalogy 84: 92-105.

Walker, C.W., Vilà, C., Landa, A., Lindén, M. and Ellegren, H. 2001. Genetic variation and population structure in Scandinavian Wolverine (Gulo gulo) populations. Molecular Ecology 10: 53-63.

3.4 European bison (*Bison bonasus bonasus*) in Poland (continental population)

Step 1.1 - Biology of the species
European bison are the largest native herbivores in the European continent. They are very similar to North American bison although they are smaller in stature and have a longer mane on the forehead and beneath the neck. The coat varies from golden brown to dark brown. European bison reach a length of about 2.9 meters and a height of about 1.8 to 1.95 meters. The weight typically goes from 800 to 1,000 kg (NA, 2012).

In the pre-rutting period (July through October), reproductively active bulls are rarely present in the group, and in July, reproductively active bulls join mixed herds with cows. Smaller groups consist of up to 12 individuals, typically including only one adult bull sexually mature (> 6 years of age) and 1 to 3 bulls that do not take part in reproduction (Krasinski and Raczynski, 1967). The gestation period typically lasts 254 to 277 days (264 days on average). European bison give birth to one calf at a time, twins are rarely observed in populations in reserves. The calving period usually occurs from May through July. Bulls in reserves reach sexual maturity in the second year of their life, while free-ranging bulls usually reach sexual maturity in the third year of life. The breeding activity in bulls typically lasts from the 6th to the 12th year of life. Females can give birth until the end of life although (but usually no more than 15 years; Krasinski and Raczynski, 1967; Pucek *et al.*, 2003).

Step 1.2 - Spatial scale of functioning
European bison was historically distributed throughout western, central, and south-eastern Europe and the Caucasus (Figure 3.4.1). By the end of the 19th century, there were only two populations of European bison left in the wild, one in Białowieża Forest in Poland (subspecies *Bison bonasus bonasus*), and one in the western Caucasus mountains (subspecies *Bison bonasus caucasicus*). *B. b. bonasus* was finally driven Extinct in the wild in 1919, and *B. b. caucasicus* in 1927. Individual home range size varies from 44 to 84 km^2.

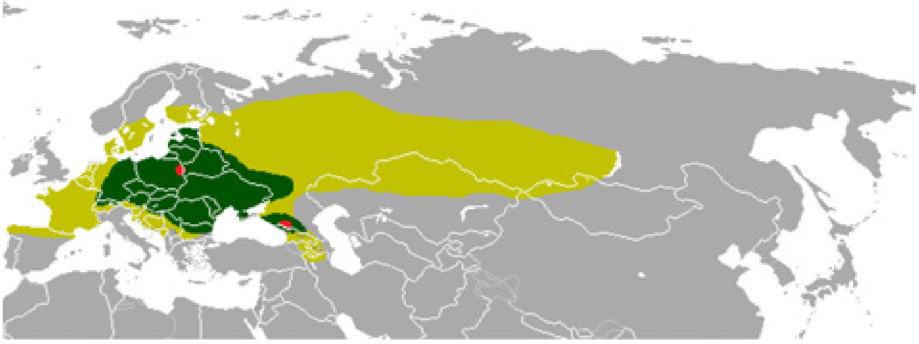


Figure 3.4.1 *Changes in the distribution of the European bison. Light green represents the range of the species in the Holocene, dark green shows the range of the bison in the late Middle Ages, while the areas in red include the range of the species in the XX century (from http://carnivoraforum.com)*

Conclusions

This species is a good example of population category S2: species with clearly disjunct distributions and one or a few isolated (often genetically differentiated) populations at the national level, because there are only two isolated populations left.

Step 1.3 - Historical perspective: what happened to the species?
Historically, European bison's range encompassed all lowlands of Europe, extending from the Massif Central to the Volga River and the Caucasus, where the species was widespread and presumably abundant (Pucek *et al.* 2004). The first population to be extirpated was that of Gaul in the 8th century

AD. The species became extinct in northern Sweden in the 11th century, and southern England in the 12th. The species survived in the Ardennes and the Vosges until the 15th century. European bison survived in a few natural forests in Europe but its numbers decreased. The last individuals in Transylvania died in 1790. In Poland, in the Białowieża Forest, wild European bison herds also existed until the mid-17th century. Polish kings took measures to protect the species, whith King Sigismund II Augustus instituting the death penalty for poaching a European bison in Bialowieza in the mid-16th century. In the early 19th century, Russian czars retained old Polish laws protecting the European bison herd in Bialowieza, but despite these measures, the European bison population continued to decline over the following century, with only Bialowieza and Northern Caucasus populations surviving into the 20th century. During World War I, German troops killed 600 of the European bisons in the Bialowieza Forest for recreation, food, hides, and horns, and at the very end of the war, retreating German soldiers shot all but 9 animals. The last wild European bison in Poland was killed in 1919, and the last wild European bison in the world was killed by poachers in 1927 in the western Caucasus. By that year the captive population consisted of just 54 (29 males, 25 females; Raczyński 1978, Pucek 1991). The captive population survived only in a few European zoological gardens (Sztolcman 1924) and subsequently increased slowly until World War II, but then the species suffered a steep decline, with the population dropping from 160 animals in 1943 to 93 in 1946 (Pucek *et al.* 2004).

After a series of successful reintroductions and introductions, European bison (subsp. *bonasus*) now occurs in isolated free-ranging and semi-free herds in Poland, Lithuania, Belarus, Russian Federation, Ukraine, and Slovakia. There was also an introduced Kyrgyzstan subpopulation, which has recently gone extinct (EBPB 1996, Pucek *et al.* 2004). Captive populations occur in 30 different countries worldwide (see Pucek *et al.* 2004 for details).

Step 1.4 - Analysis of distribution and trends

As a result of captive breeding, reintroductions, benign introductions, and intensive conservation management, the total population of free-ranging bison in continental Poland now stands at c. 835, and additional 1,400 individuals live in captivity (EBPB 2004). Approximately 60% of individuals are sexually mature (Krasiński 1978, Krasińska and Krasiński 2004). The free-living population increased more or less steadily from the mid-1960s to a peak of c. 2000 in the early 1990s (Pucek *et al.* 2004). Following a period of decline in the mid to late 1990s, the population is now once again expanding (Olech 2008). Considering these data, we conclude that the population size of the European bison has increased steadily in the last 50 years, despite a period of decline in the second half of the 1990s.

Conclusions

The European bison (Bison b. bonasus) has experienced a steep decline in both its geographical range and population size with respect to historical times, since it was considered extinct in the Wild in 1927. Since 1960, as a result of captive breeding, reintroductions, benign introductions, the free-ranging population of the bison has been increasing, and it is still expanding in Poland. Although both the distribution and population size of the species have increased in the last 50 years, they were larger in the historical past.

Step 2.1 - FRP assessment

Apparently the European bison (subsp. *bonasus*) is still expanding within its historical natural range. We propose that the FRP for the population in Poland (CON) should be in the order of magnitude of thousands of individuals (874 free-ranging individuals reported in 2013 for PL CON). This is because only large (1,000-3,000 individuals) populations can prevent the species from extinction due to genetic bottlenecks and inbreeding depression (Pucek 2004).

Since the survivor and expansion of the population of the species has been strictly related to conservation measures, instead of setting this number as the FRP, we use operators and suggest to constantly monitor the status of the population until range and population size are stable for a sufficient long time. This would be the final FRP value (and see step 2.2 FRR assessment).

Conclusions

For PL CON FRP > 1000.

Step 2.2 - FRR assessment

Based on the previous discussion on the extinction in the wild of the European bison, and the fact that it is still expanding, we suggest a FRR>CV (in 2013 reported as 3436 for PL CON). However, the potential for ongoing growth may limited by a number of factors related to the human interference (e.g., habitat loss and degradation, fragmentation of populations and concomitant loss of genetic diversity, inbreeding depression, disease, hybridisation, and poaching; Pucek *et al.* 2004).

Conclusions

For PL CON FRR > CV.

References

European Bison Pedigree Book. 1996. European Bison Pedigree Book. Białowieża National Park.

Krasinski, Z., J. Raczynski. 1967. The reproduction biology of European bison living in reserves and in freedom. Acta Theriologica, 12: 407-444.

Krasiński, Z. A. 1978. Dynamics and structure of the European bison population in the Białowieża Primeval Forest. Acta Theriologica 23: 3-48.

Krasińska, M. and Krasiński, Z. A. 2004. [European bison. Natural Monograph]. ZHUBR. Monografia przyrodnicza. Studium Fotografii Przyrodniczej HAJSTRA. [In Polish]. Warszawa-Białowieża: 1-312.

NA, 2012. "About European Bison". European Bison Conservation Center. Accessed August 10th 2017 at http://www.bison-ebcc.eu/about-european-bison/.

Olech, W. (IUCN SSC Bison Specialist Group). 2008. Bison bonasus. The IUCN Red List of Threatened Species 2008: e.T2814A9484719. http://dx.doi.org/10.2305/IUCN.UK.2008.RLTS.T2814A9484719.en. Downloaded on 21 August 2017.

Pucek, Z. 1991. History of the European bison and problems of its protection and management. In: B. Bobek, K. Perzanowski and W. Regelin (eds), Global Trends in Wildlife Management, pp. 19–39. Świat Press, Kraków-Warsaw, Poland.

Pucek, Z., I. Belousova, Z. Krasinski, M. Krasinska, W. Olech. 2003. European bison (Bison bonasus) Current State of the species and an action plan for its conservation. Convention on the Conservation of European Wildlife and Natural Habitats, 20: 28-29.

Pucek, Z. 2004. European Bison. Status survey and Conservation Action Plan. IUCN/SSC Bison Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Raczyński, J. 1978. Żubr [European Bison]. Państwowe Wydawnictwo Rolnicze i Leśne, Warsaw, Poland.

Sztolcman, J. 1924. Matériaux pour l'histoire naturelle et pour l'historique du Bison d'Europe (Bison bonasus Linn.). Annales Zoologici Musei Polonici Historiae Naturalis.

3.5 Eurasian lynx (*Lynx lynx*) in the Alps

Step 1.1 - Biology of the species

The Eurasian lynx (*Lynx lynx*) is the largest among all the four lynx species as well as. one of the largest predators in Europe, only exceeded by brown bears and grey wolves. Their body mass ranges from 18 to 36 kg, with males being larger and more robust than females (Nowell & Jackson 1996). Typical features of the Eurasian lynx are black tufts at the tips of the ears and a prominently flared facial ruff. The mating season starts in February until the end of April, but each female is fertile only about three days during this period. Once the female is no longer in estrus, the male leaves to find another mate. Females have only one mate per season (polygynous mating system; Nowell & Jackson 1996).

Gestation period lasts 67 to 74 days, with females giving birth in May. Breeding interval varies, depending on success of the previous season. Females without a litter can breed every year, while females with a litter can breed about every 3 years. Litter size varies from 1 to 5 kittens (usually 2 or 3). Newborn cubs weaned at 4 months and become independent at around 10 months. Females reach sexual maturity at 2 years of age and can remain so up to 14 years of age, whereas males become sexually mature at 3 years of age and can reproduce up to age 17 (Nowell & Jackson 1996).

Step 1.2 - Spatial scale of functioning

The Alpine population is surveyed by camera-trapping (including capture-mark-recapture in reference areas and density extrapolation) combined with the collection of different data sets validated using the criteria developed by the Status and Conservation of the Alpine Lynx Population (SCALP) project (Molinari-Jobin *et al.* 2003, Molinari-Jobin *et al.* 2012). The home range of individual lynxes is estimated as 71–281 km^2 (Breitenmoser *et al.* 1993). Alpine lynxes are considered as one supra-national population despite the fact that (after re-introduction) several relatively isolated subpopulations still occur (see Step 1.4).

Conclusions

Since the lynx has a large home range shared between Switzerland, Slovenia, France, Italy and Austria, this species in the Alps is a good example of population category S4: animal species with individuals with large home ranges (>100 km^2 up to >1000 km^2).

Step 1.3 - Historical perspective: what happened to the species?

In historical times, the lynx existed throughout Europe with the exception of the Iberian Peninsula, most Islands, un-forested coastal regions, and the north-west of northern Europe. In the Alps, the expansion of human population and the over-exploitation of natural habitats and resources, including forests and game, led to the historic decline and eradication of the large carnivores between 1800 and the early 1900. An increase in livestock negatively affected forests 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, the main driver of extinction for the large carnivores, especially lynx and wolf, was the massive intervention at the level of the landscape (forests) and the substantial reduction of wild ungulates (Zimen 1978, Breitenmoser 1998a). Only in the first half of the 20th century a radical change in forest management and the growing sensitivity of people for the protection of nature put the basis for the recovery of the forests (Breitenmoser 1998a). Wild ungulates started to recover and expand after they were granted a certain legal protection (change of hunting legislation), with a consequent and continuing increase in the populations of roe deer, red deer and wild boar in many regions. This ecological recovery was facilitated by industrialisation, which drew people away from rural areas, which led to a drastic decline in the number of goats and sheep in the Alps in the first half of the 20th century. All these factors prepared the ground for the return of lynx to the Alps. Today, the Alpine population of the lynx consists of several occurrences, all originating from reintroductions in the 1970s: Switzerland 1970–76 (Breitenmoser *et al.* 1998), in the Dinaric part of Slovenia 1973 (Cop & Frkovic 1998) and Austria 1977–79 (Huber & Kaczensky 1998). The most important population is currently found in the north-western Alps of Switzerland. Although lynx immigrated into neighbouring countries (France and Italy), the population throughout the Alps is not continuous.

Kaczensky *et al.* (2013) assessed the large carnivore populations across Europe, based on the most recent data available, and the Alpine lynx population has been evaluated as Endangered under the IUCN 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 decreasing in Slovenia. The authors of the assessment concluded 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".

Step 1.4 - Analysis of distribution and trends

The current Alpine lynx population originates from reintroductions in the 1970s (Schnidrig *et al.* 2016). Two main populations can be identified: one in the western to central Swiss Alps (cantons of Valais, Vaud, Fribourg and Berne), stretching into the French Alps, and the western Alps in Austria (Vorarlberg), and another one in the western part of Slovenia (west of the Jesenice-Ljubljana-Triest highway), stretching into the adjacent regions of Italy (Tarvisiano) and Austria (Carinthia; Figure 3.5.1; Molinari-Jobin 2010). In addition, areas with the a more scattered lynx distribution are found in France (south-east of the country, from the lake of Geneva as far south as to the department of Hautes-Alpes), eastern Italy (Friuli Venezia Giulia, Veneto (Bellunese), and Austria (northern Kalkalpen, Upper Carinthia, Niedere Tauern), with no permanent lynx presence (von Ark 2007).

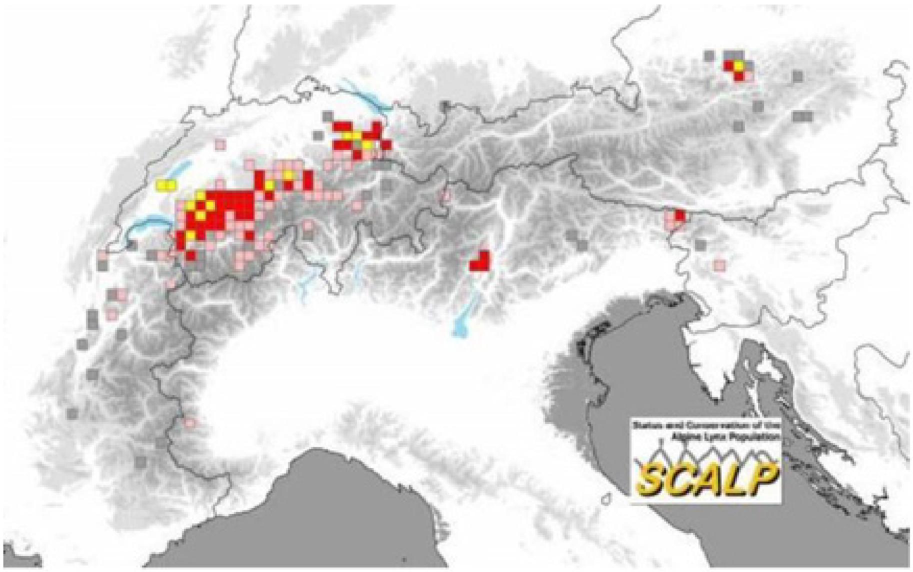


Figure 3.5.1 *Distribution of the Eurasian lynx in the Alps in 2012 (from SCALP 2012). red cells = C1, cells with hard fact records, pink cells = C2, cells with confirmed records, grey cells = C3, cells with unconfirmed record*

Over the past 10 years, lynxes have increased their range by around 6,000 km^2 or 50%, mainly after a translocation in Switzerland. However, the population size remained more or less the same. In 2011, the number of lynx estimated for the entire Alpine region was between 130 and 180 individuals (Schnidrig *et al.* 2016)(Table 3.5.1). 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 they show today a high inbreeding coefficient (Schnidrig *et al.* 2016. Reproduction has been reported in only 4 areas: north-western Swiss Alps, Friuli, north-eastern Switzerland and the Chartreuse region (Molinari Jobin *et al.* 2010a, b). Recently, reproduction also occurred in the Kalkalpen (Fuxjäger 2014).

***Table 3.5.1** Lynx abundance in the Alps (from Schnidrig et al. 2016)*

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	
Total Alps	**~120**	**136–180**	**~130**	

According to Schnidrig *et al.* (2016) the "French Alpine sub-population was assessed to be stable with 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 Italian Alps were naturally recolonised in the 1980s by individuals from the reintroduced populations. However, the initial positive trend was soon reversed and population estimates decreased from 21 individuals in 1995 with observed reproduction (Ragni *et al.* 1998) down to less than 15 in 2009 with no signs of reproduction (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).

During 2000–2004 a total of 30–50 lynx were estimated for Slovenia, of which 15 in the Alpine part (Koren *et al.* 2006). 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 *et al.* 2012). In addition, no reproduction had ever been recorded in the Slovenian Alps up to then (Potočnik *et al.* 2009). The population of lynx in the Alpine region has further decreased to few (2-3) individuals.

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. The population in Switzerland currently forms the largest subpopulation of lynx in the Alpine region (von Arx & Zimmermann 2013) with an estimated population of 96–107 independent individuals (von Arx & Zimmermann 2013)."

Conclusions

The Alpine population of the lynx was completely extirpated from the Alpine territory about one century ago. Despite reintroductions have been successful in some countries (especially in Switzerland, where the most important population currently occurs), and the fact that over the past 10 years, they have increased their range by around 6,000 km2 or 50%, mainly after a translocation in Switzerland, the population remained very small, threatened and fragmented.

Step 2.1 - FRP assessment

Thanks to reintroductions, the species has re-colonized some portions of its historical range in the Alps. It is unlikely that the eastern and western Alpine subpopulations will merge in the coming decades. Despite estimates of population and range size for the period before the extinction of the Alpine population are very scarce, we know that the species was widespread throughout the Alps in historical times. Major threats include inbreeding depression, illegal killing, infrastructure development (especially road constructions), vehicle and train collisions, and limited dispersal (von Arx 2007), but the species might still be able to re-colonize much of its historical range if law enforcement improves, monitoring of demographic and genetic parameters continues, and acceptance of lynx by local people increases (von Arx 2007). Therefore, we should use operators to suggest that FRP is much larger than current value.

Conclusions

For the lynx population in the Alps (as defined in section 1.4) FRP >> CV.

Step 2.2 - FRR assessment

Based on the previous discussion on the increase in distribution of the Alpine population in the last 15 years and considering variation within the natural range, we conclude that the range can still increase (i.e. likely to be much higher than the current 24,300 km^2 including permanent and sporadic ranges; Chapron *et al.* 2014), if major threats to the species are reduced.

Conclusions

For the lynx population in the Alps (as defined in section 1.4) FRR >> CV.

References

Breitenmoser, U., Kavczensky, P., Dötterer, M., Breitenmoser-Würsten, C., Capt, S., Bernhart, F., & Liberek, M. (1993). Spatial organization and recruitment of lynx (Lynx lynx) in a re-introduced population in the Swiss Jura Mountains. Journal of Zoology, 231(3), 449-464.

Breitenmoser, U. 1998a. Large predators in the Alps: the fall and rise of man’s competitors. Biol. Conserv. 83(3), 279-289.

Breitenmoser, U., Breitenmoser-Würsten, Ch. & Capt S. 1998. Re-introduction and present status of the lynx in Switzerland. Hystrix 10(1), 17-30.

Cop, J. & Frkovic, A. 1998. The re-introduction of the lynx in Slovenia and its present status in Slovenia and Croatia. Hystrix 10(1), 65-76.

Fuxjäger, C. 2014. Luchsmonitoring 2013 – Jahresbericht. Natinalpark Kalkalpen, Molln, Austria, 10 pp.

Huber, T. & Kaczensky, P.1998. The situation of the lynx in Austria. Hystrix, 10(1):43-54.

Kaczensky, P., Chapron, G., von Arx, M., Huber, D., Andrén, H. & Linnell, J. (Eds) 2013. Status, management and distribution of large carnivores - bear, lynx, wolf & wolverine - in Europe. Part I. Europe summaries. A Large Carnivore Initiative for Europe Report prepared for the European Commission (contract N°070307/2012/ 629085/SER/B3). 72 pp.

Koren, I., Jonozovic, M. & Kos, I. 2006. Status and distribution of the Eurasian lynx (Lynx lynx L.) in Slovenia in 2000-2004 and comparison with the years 1995-1999. Acta Biologica Slovenica 49(1), 27-41.

Kos, I. & Potočnik, H. 2013. Lynx- Slovenia. In Kaczensky P., Chapron G., von Arx M., Huber D., Andren H. & Linnell J. 2013. Status, management and distribution of large carnivores- bear, lynx, wolf & wolverine- in Europe. Part 2. European Commission.

Molinari-Jobin, A. *et al.* 2003. Pan-Alpine conservation strategy for the lynx. Nature and Environment No. 130 (Council of Europe Publishing, Strasbourg, France), pp. 1–25.

Molinari-Jobin, A., Marboutin, E., Wölfl, S., Wölfl M., Molinari, P., Fasel, M., Kos, I., Blazic, M., Breitenmoser-Würsten, Ch., Fuxjäger, Ch., Huber, T., Izotok, K. & Breitenmoser, U. 2010a. Recovery of the Alpine lynx Lynx lynx metapopulation. Oryx 44 (2), 267–275.

Molinari-Jobin, A., Kos, I., Marboutin, E., Molinari, P., Wölfl, S., Fasel, N., Breitenmoser-Würsten, C., Fuxjäger, C., Huber, T., Koren, I., Schmidt K., Kusak, J., Valdmann, H., Zimmermann, F., Wölfl, M. & Breitenmoser, U. 2010b. Expansion of lynx in the Alps. KORA and Bavarian Environment Agency. KORA Bericht Nr. 50. 17 pp.

Molinari-Jobin, A. *et al.* 2012 Monitoring the lynx in the Alps. Hystrix 23, 49–53.

Nowell, K., Jackson, P. 1996. Wild Cats: Status survey and conservation action plan. Cambridge, U.K.: IUCN: The Burlington Press.

Potočnik, H., Skrbinšek, T. & Kos, I. 2009. The reintroduced Dinaric lynx population dynamics in PVA simulation, The 30 years retrospection and the future viability. Acta Biologica Slovneica 52(19) 3-18.

Ragni, B., Possenti, M., Mayr, S., Carrer, M., Zangrando, E., Catello, M., Dorigatti, E., Di Lorenzo, M., Mosca, A., Fattor, M. & Lombardi, G. 1998. The lynx in the Italian Alps. Hystrix 10(1), 31-38.

SCALP. 2012. Monitoring Report 2009-2011. 3pp.

Schnidrig, R., Nienhuis, C., Imhof, R., Bürki, R., & Breitenmoser, U. (Eds). 2016. Lynx in the Alps: Recommendations for an internationally coordinated management. RowAlps Report Objective 3. KORA Bericht Nr. 71. KORA, Muri bei Bern, Switzerland, and BAFU, Ittigen, Switzerland, 70 pp.

von Arx, M. (Large Carnivore Initiative for Europe / Cat Specialist Group). 2007. Lynx lynx. The IUCN Red List of Threatened Species 2007: e.T12519A3351478. Downloaded on 06 September 2017.

von Arx, M. & Zimmermann, F. 2013. Lynx – Switzerland. In Status, management and distribution of large carnivores - bear, lynx, wolf & wolverine - in Europe. Part II. Kaczensky P., Chapron G., von Arx M., Huber D., Andrén H. & Linnell J. (Eds). A Large Carnivore Initiative for Europe Report prepared for the European Commission.

Zimen, E. 1978. Der Wolf. Mythos und Verhalten. Meyster Verlag GmbH, Wien, München. 333pp.

3.6 Apennine brown bear (*Ursus arctos marsicanus*) in the Italian Apennines

with the support of Dr. Paolo Ciucci

Step 1.1 - Biology of the species
The small brown bear population in the Italian Apennines (*Ursus arctos Apennine brown bear*) was characterized by a prolonged period of isolation (at least 400-600 years) which resulted in significant genetic (Randi *et al.* 1994) and morphological differentiation (Loy *et al.* 2008, Colangelo *et al.* 2012) from the bears living in the Alps and in the rest of Europe. Therefore, the remaining Apennine population of brown bears should be regarded as an evolutionary and conservative unit of self-preservation (AA.VV. 2009, Colangelo *et al.* 2012).

The males of the Apennine brown bears can weigh up to 200 kg, with an upright height of 1.9 to 2 metres, while females are generally smaller (140 kg; Loy *et al.* 2008). The individuals are usually very shy, and appear only at night. Most of the bears are solitary and occupy their own territory, which can be up to 300 km^2 in reproductive males.

The bear shows a strong preference for mountain forest ecosystems (oak trees, beech trees and coniferous forests) within which it carries out most of its activities. However, this may be, at least in part, a behavioural response to anthropic impact. The median altitude distributional range is very wide and varies with the season, especially in relation to the availability of food resources (Liste Rosse italiane 2013).

The brown bear is omnivorous. The vegetable component, including herbs and fruits, is prevalent in all seasons, although protein sources (mammals, insects) are also frequently used. Experts found that the diet of the bear varies seasonally (Ciucci *et al.* 2014). Hard mast production strongly influenced year-to-year variation in the diet. High-quality foods, such as berries and other fleshy fruits, were increasingly consumed by bears in years of low to null hard mast productivity. This diversified and abundant habitat productivity should avoid the risk of nutritional stress during occasional hard mast failures (Ciucci *et al.* 2014).

Breeding generally occurs in late spring or early summer, but mating outside this breeding season has been also reported, with observations made also in autumn (Tosoni at al. 2011). Females usually give birth to 1-3 cubs, who spend about 1.5 years with their mother (Tosoni *et al.* 2017a). The females are very philopatric (P. Ciucci pers. comm. 2017). From 3 to 7 reproducing females have been estimated each year in the population, and their inter-birth interval is 4-5 years (Tosoni *et al.* 2017a, 2017b).

Step 1.2 - Spatial scale of functioning
The range of the Apennine brown bear mainly develops within the PNALM (Figure 3.6.1, Abruzzo National Park, Lazio and Molise) which covers an area of 1300 km^2 (mean density here 33 individuals/1000 km^2; Ciucci & Boitani 2008, 40 individuals/1000 km^2 in the core area of the park; Ciucci *et al.* 2015). In the areas outside the park, the presence of erratic individuals is sporadic (e.g. Simbruini, Ernici, Gran Sasso and Maiella National Parks), and they are present in extremely low densities (Bosagli 1999, Ciucci & Boitani 2008).

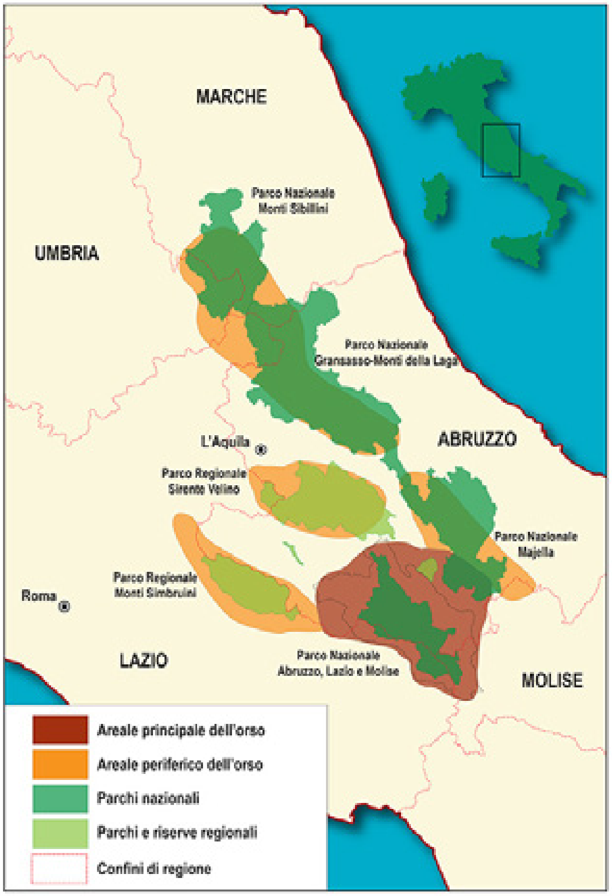


Figure 3.6.1 *Apennine (or Marsican) brown bear distribution in the Apennines. Brown polygons: core range of the bear. Orange polygons: : peripheral range. Dark green: National Parks. Light green: Regional Parks and Reserves. Dotted red lines indicate regional boundaries (from http://www.parcoabruzzo.it)*

Conclusions

The Apennine brown bear is a good example of population category S2: species with clearly disjunct distributions and one or a few isolated (often genetically differentiated) populations at the national level.

Step 1.3 - Historical perspective: what happened to the species?

The Apennine brown bear population has been separated from the Alpine population for at least 400-600 years (Randi *et al.*, 1994). Following random drift and extinction of maternal lineages since isolation, the original genetic diversity of this small brown bear population has been severely depleted (Lorenzini *et al.* 2004) and the population now has a unique mtDNA haplotype (Randi *et al.* 1994). The first formal estimate of the size of the Apennine brown bear population was produced in 2008, indicating 40 (95% *CI* = 37–52) bears in the core range (Gervasi *et al.* 2012).

Historically the Apennine bear distribution spanned most of the Apennine range (Zunino and Herrero 1972, Carpaneto and Boitani 2003) but it has increasingly been reduced since the 17th century (Febbo and Pellegrini 1990, Boscagli *et al.* 1995). Consistent range reductions have occurred, in particular, over the past 200 years, mostly due to human persecution (Febbo and Pellegrini 1990). In 1923, the PNALM was formally established, and by the 1970s the remnant bear population survived almost exclusively in the Park and its immediate surrounding mountains (Zunino and Herrero 1972). The core of the Apennine brown bear population range includes the PNALM ecosystem (the Park and its contiguous areas, for about 1,500-2,500 km^2), whereas the peripheral area includes a limited number of bears, possibly occasional dispersers. This marginal population is partly included within a larger network of protected areas in the Central Apennines (Figure 3.6.1), in a matrix of more than

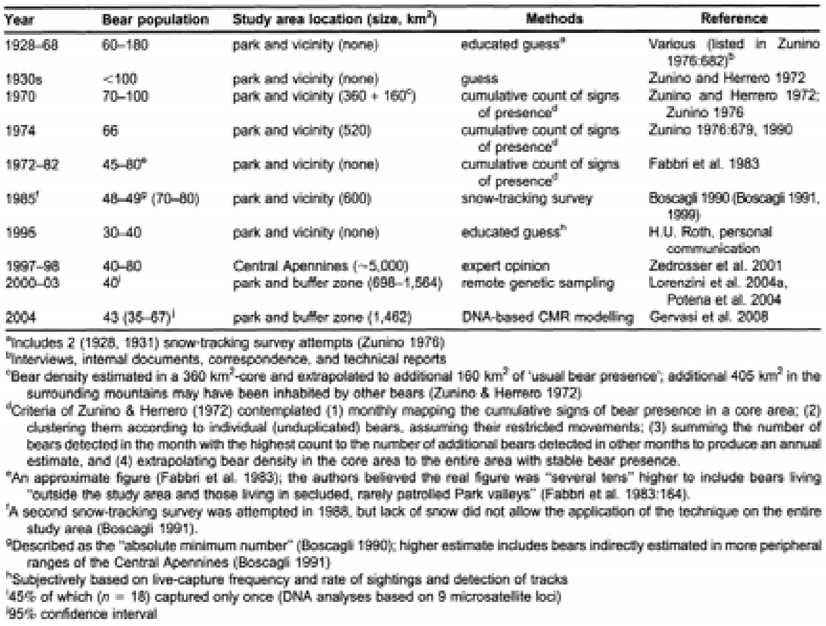
10,000 km^2 of estimated suitable bear habitat (Falcucci *et al.* 2007). However, In this area high levels of human activity might provide excessive disturbance to bears, thus creating a significant gap in connectivity between suitable habitat patches (Posillico *et al.* 2004, Falcucci *et al.* 2009). Although no formal method has been applied to assess its range, in the last 50 years it is believed that the range of the species has remained mostly stable (P. Ciucci pers. comm. 2017).

Step 1.4 - Analysis of distribution and trends

During the years 1980-1985 there has been an increase in the extent of the PNALM park and in the number of wild boars, with a consequent increase in hunting and poaching. This caused collateral damages to bears, and it is probably the period with the highest number of dead bears (10-11 adult bears, mostly males, died in 1985; P. Ciucci pers. comm. 2017). In the mid-1990s, based on frequency of live captures and detection of tracks and sightings as part of a radiotelemetry bear research, about 30-40 bears were believed to be living in the Park (Ciucci & Boitani 2008). The population in the last 50 years has remained mostly stable, with oscillations between 40 and 60 individuals (P. Ciucci pers. comm. 2017). A population size of 40-50 individuals and a range of 1800 km^2 have been reported in 2013 for IT MED. No increases have been reported (see Table 3.6.1 for details).

Considering these data, we conclude that the population size of the Apennine brown bear has remained mostly stable in the last 50 years. Likely there is an elevated cub mortality during their first year of life (Gervasi *et al.* in press). It is not known if that is due to inbreeding depression or to the fact that the animals disperse outside the core zone in areas that are not safe (i.e., habitat sinks). The effective genetic population is composed by 16-17 individuals, therefore it is (and has been) highly subject to loss of genetic variability (P. Ciucci pers. comm. 2017).

Table 3.6.1 *Historical population numbers of Apennine brown bear*



Year	Bear population	Study area location (size, km^2)	Methods	Reference
1928–68	60–180	park and vicinity (none)	educated guess[a]	Various (listed in Zunino 1976:682)[b]
1930s	<100	park and vicinity (none)	guess	Zunino and Herrero 1972
1970	70–100	park and vicinity (360 + 160[c])	cumulative count of signs of presence[d]	Zunino and Herrero 1972; Zunino 1976
1974	66	park and vicinity (520)	cumulative count of signs of presence[d]	Zunino 1976:679, 1990
1972–82	45–60[e]	park and vicinity (none)	cumulative count of signs of presence[d]	Fabbri et al. 1983
1985[f]	48–49[g] (70–80)	park and vicinity (600)	snow-tracking survey	Boscagli 1990 (Boscagli 1991, 1999)
1995	30–40	park and vicinity (none)	educated guess[h]	H.U. Roth, personal communication
1997–98	40–80	Central Apennines (~5,000)	expert opinion	Zedrosser et al. 2001
2000–03	40[i]	park and buffer zone (698–1,564)	remote genetic sampling	Lorenzini et al. 2004a, Potena et al. 2004
2004	43 (35–67)[j]	park and buffer zone (1,462)	DNA-based CMR modelling	Gervasi et al. 2008

[a] Includes 2 (1928, 1931) snow-tracking survey attempts (Zunino 1976)

[b] Interviews, internal documents, correspondence, and technical reports

[c] Bear density estimated in a 360 km^2-core and extrapolated to additional 160 km^2 of 'usual bear presence'; additional 405 km^2 in the surrounding mountains may have been inhabited by other bears (Zunino & Herrero 1972)

[d] Criteria of Zunino & Herrero (1972) contemplated (1) monthly mapping the cumulative signs of bear presence in a core area; (2) clustering them according to individual (unduplicated) bears, assuming their restricted movements; (3) summing the number of bears detected in the month with the highest count to the number of additional bears detected in other months to produce an annual estimate, and (4) extrapolating bear density in the core area to the entire area with stable bear presence.

[e] An approximate figure (Fabbri et al. 1983); the authors believed the real figure was "several tens" higher to include bears living "outside the study area and those living in secluded, rarely patrolled Park valleys" (Fabbri et al. 1983:164).

[f] A second snow-tracking survey was attempted in 1988, but lack of snow did not allow the application of the technique on the entire study area (Boscagli 1991).

[g] Described as the "absolute minimum number" (Boscagli 1990); higher estimate includes bears indirectly estimated in more peripheral ranges of the Central Apennines (Boscagli 1991)

[h] Subjectively based on live-capture frequency and rate of sightings and detection of tracks

[i] 45% of which ($n = 18$) captured only once (DNA analyses based on 9 microsatellite loci)

[j] 95% confidence interval

Conclusions

The Apennine brown bear has experienced a decline in its geographical range with in the historical past. It is believed that it has also experienced a reduction in population size, although estimates before 2008 are not reliable. In general, it is most plausible that both the population and range size have remained relatively stable in the last 50 years. Despite this stability, the species is at high risk of loss of genetic variability and consequent inbreeding depression.

Step 2.1 - FRP assessment

Given the assumed relative stability of the population in the last 50 years, and the past and current risk of inbreeding depression to which this population has been subject, the FRP value should be at least >50 reproductive individuals, which is the minimum recommended population size to minimize damages due to the deleterious effect of inbreeding only. Based on a Ne to N ratio of about 0.30 (P. Ciucci pers. comm. 2017), the ideal total population should then correspond to 250-260 bears, and the recently modelled environmental suitability at the Apennine scale could theoretically accommodate for such a population size (P. Ciucci pers. comm. 2017). This would imply a remarkable extension of the geographic range, well beyond the current borders.

Conclusions

For IT MED FRP is about 250 individuals.

Step 2.2 - FRR assessment

Based on the previous discussion on the ongoing decrease in the current distribution, and the fact that potentially suitable habitat exists outside the core area, we would suggest a FRR>DV. However, when projecting both an increase in population and range size, interference with human land use and safety considerations should be necessarily considered in the planning.

Conclusions

For IT MED FRR > CV (1800).

References

AA VV (2009), Piano d'Azione per la Conservazione dell'Orso Apennine brown - PATOM. Documenti Tecnici.

Boscagli, G. 1987. Brown bear mortality in Central Italy from 1970 to 1984. International Conference Bear Research and Management 7:97-98.

Carpaneto, G.M., & Boitani, L. 2003. Ursus arctos distribuzione geografica. Pages 92-94 in L. Boitani, S. Lovari, and A. Vigna Taglianti, editors. Fauna d'ltalia, Vol. XXXVII. Mammalia. III. Carnivora?Artiodac tyla. Calderini, Bologna, Italy. (In Italian.)

Ciucci, P., & Boitani, L. (2008). The Apennine brown bear: a critical review of its status and conservation problems. Ursus, 19(2), 130-145.

Ciucci, P., Tosoni, E., Di Domenico, G., Quattrociocchi, F., & Boitani, L. (2014). Seasonal and annual variation in the food habits of Apennine brown bears, central Italy. Journal of Mammalogy, 95(3), 572-586.

Ciucci, P., Gervasi, V., Boitani, L., Boulanger, J., Paetkau, D., Prive, R. & Tosoni, E. (2015). Estimating abundance of the remnant Apennine brown bear population using multiple noninvasive genetic data sources. J Mammal; 96 (1): 206-220. doi: 10.1093/jmammal/gyu029

Colangelo, P., Loy, A., Huber, D., Gomercic, T., Vigna Taglianti, A., & Ciucci, P. (2012), Cranial distinctiveness in the Apennin brown bear: genetic drift effect or ecophenotypic adaptation? Biological Journal of the Linnean Society n.107 (1): pp. 15-26.

Falcucci, A., Ciucci, P., Maiorano, L., Gentile, L. and Boitani, L. (2009), Assessing habitat quality for conservation using an integrated occurrence-mortality model. Journal of Applied Ecology, 46: 600–609. doi:10.1111/j.1365-2664.2009.01634.x

Febbo, D., & Pellegrini, M. (1990). The historical presence of the brown bear in the Apennines. Aquilo Serie Zool?gica 27:85-88.

Gervasi, V., Ciucci, P., Boulanger, J., Randi, E., Boitani, L. (2012). A multiple data source approach to improve abundance estimates of small populations: the brown bear in the Apennines, Italy. Biological Conservation 152:10–20.

Lorenzini, R., M. Posillico, S. Lovari, and A. Petrella. (2004). Noninvasive genotyping of the endangered Apennine brown bear: A case study not to let one’s hair down. Animal Conservation 7:199-209.

Loy, A.; P. Genov; M. Galfoc; M. G. Jacobonec; A. Vigna Taglianti (2008), “Cranial morphometrics of the Apennine brown bear (Ursus arctos Apennine brownus) and preliminary notes on the relationships with other southern European populations”, Italian Journal of Zoology, 75 (1): 67–75.

Randi, E., Boscagli, G., Huber, D. & Roth, H. (1994), Mitochondrial DNA sequence divergence among some west european brown bear populations Heredity pp. 480-389

Tosoni, E. (2010). Indagini ecologiche relative alla popolazione di orso bruno Apennine browno (Ursus arctos Apennine brownus) nel territorio del Parco Nazionale d’Abruzzo, Lazio e Molise. PhD Thesis.

Tosoni, E., Paolozzi, G., Trella, E., Boitani, L., & Ciucci, P. (2012). A case of autumn mating in the Apennine brown Bear (Ursus arctos Apennine brownus). Hystrix, the Italian Journal of Mammalogy, 22(2).

Zunino, F., & Herrero, S. (1972). The status of the brown bear in Abruzzo National Park, Italy, 1971. Biological Conservation 4:263-272.

3.7 Wolf (*Canis lupus*) in the Iberian Peninsula (NW Iberian population)

Step 1.1 - Biology of the species
The wolf (*Canis lupus*) in Europe occurs in all countries except in the Benelux, Denmark, Hungary and the island states (Ireland, Iceland, United Kingdom, Cyprus, Malta). Based on a combination of distribution and social, ecological and political factors, Chapron *et al.* (2014) categorized these into 10 populations: North Western Iberian, Sierra Morena, Alpine, Italian Peninsula, Carpathian, Dinaric-Balkan, Baltic, Karelian, Scandinavian and Central European Lowlands.

The Iberian wolf (*C. lupus signatus*) is a subspecies of gray wolf endemic to the Iberian Peninsula. This subspecies is slightly smaller than northern wolves, with distinctive white marks on the upper lips, dark marks on the tail and a pair of dark marks on its front legs that give it its subspecies the name "signatus" (which means "signed or marked"; Torres *et al.* 2016). A morphometric analysis showed differences in skull shape separating Iberian wolves from wolves in Italy and other populations in Eastern Europe (Vilà *et al.* 1993). Mitochondrial DNA data and microsatellite frequencies showed a high differentiation between Iberian wolves and those found elsewhere in Eurasia (Vilà and Wayne 1999; Lucchini *et al.* 2004). Individuals live in small packs. They eat wild boar, rabbits, roe deer, red deer, ibexes and even small carnivores and fish. The species can live up to thirteen years in the wild, though average lifespan is 5 to 6 years. Most packs are made up of 5 to 9 individuals, usually composed by an alpha pair and their offspring (http://animaldiversity.org).

Step 1.2 - Spatial scale of functioning
Studies based on telemetry indicated that the mean size of wolf territories vary between 170-240 km^2 (Boitani *et al.* 2003). The North Western population, the focus of this example, is shared between Spain and Portugal (Figure 3.7.1), where the range of the wolf extends for > 120000 km^2. The other Spanish population of wolf is located in Sierra Morena and is genetically distinct from the one in the North-West of the Iberian Peninsula. The nearest wolf population outside Iberia is in the western Alps and there are no connections between the two. However, some wolves from the Alps have been reaching the Pyrenees, although breeding has not been confirmed yet (Kaczensky *et al.* 2012).

Figure 3.7.1 *Wolf distribution in the Iberian Peninsula 2006-2011. Dark cells: permanent occurrence, Grey cells: sporadic occurrence (modified from Kaczensky et al. 2012)*

In Portugal, around 80% of the area with confirmed presence of the wolf is north of the river Duero. The Spanish population occupies the northwestern portion of the country, and is the largest one in Spain. In Spain, the Iberian wolf population covers some 120,000 km^2. More than 90% of wolves are concentrated in three autonomous regions: Castilla y León, Galicia and Asturias. In addition, the wolves also breed in 4 or 5 more autonomous regions: Cantabria, Basque Country, Castilla-La Mancha (Guadalajara province), Madrid and, perhaps, La Rioja (Kaczensky *et al.* 2012).

Conclusions

This species is a good example of population category S4 as the wolf has large home range, and forms supra-national populations in Europe (such as the NW Iberian population).

Step 1.3 - Historical perspective: what happened to the species?

The historical distribution of the wolf included almost all European countries, and in the 18th century it was still present everywhere (with the exception of UK and Ireland). In the 20th century, the wolf disappeared from central and northern Europe, and in the 1970s it was present only in Spain, Portugal, Italy, Greece, ex-Yugoslavia and Scandinavia (Boitani *et al.* 2003).

Until the 1900s the Iberian wolf inhabited the majority of the Iberian Peninsula. The species used to be widely distributed in Portugal but its population began a steady decline around 1930 (Torres *et al.* 2016). Spain’s Francoist government started an extermination campaign during the 1950s. In the time period between 1950 and 1970, there were 350-500 individuals in the Spanish part of the population, while Portugal hosted 150-200 wolves.

Step 1.4 - Analysis of distribution and trends

The size of the Iberian population has not been recently estimated. It occupies 8 autonomous regions (in addition, all the wolves of Portugal are included in the Iberian population) and every region carries out their own wolf estimates but not in a coordinated way. The most recent reliable regional surveys are as follows (the year is included in brackets): Galicia 68 packs (2003), Asturias 35 packs (2004), Cantabria 5 packs (1997), Basque Country 2 packs (2010), Castilla y León 149 packs (2001), Castilla-La Mancha (Guadalajara province) 2 packs (2011), Madrid 1 pack (2011; Kaczensky *et al.* 2012). Studies conducted in 2010, 2011 and 2012 indicated an increase in the number of individuals, with ~2000 individuals in NW Spain (Chapron *et al.* 2014), although the population has been expanding since 1970 but has been rather stable since 2002 (Kaczensky *et al.* 2012). In the national census of Portugal carried out in 2002/2003, 63 wolf packs were found (51 confirmed and 12 probable), i.e., approximately 220-430 individuals. However, recent studies (2009/2010) conducted in specific areas (Trás-os-Montes area, South Douro river area) reveal a decrease in the number and reproductive stability of breeding packs (Kaczensky *et al.* 2012). Member states in 2013 indicated a population of 92-114 individuals for Portugal (PT ATL + PT MED) and 1850-2800 individuals for Spain (ES ATL + ES MED).

With respect to the geographic range of this population, in the period 1950-1970 the total range of the Iberian population was estimated at 69200 km^2. In recent years (2010-2012), the range of the NW Iberian population has reached 120300 km^2 (116000 km^2 permanent range and 3700 km^2 sporadic range; Chapron *et al.* 2014).

Based on these data, we conclude that the population size of the NW Iberian population has increased threefold in the period 1950-2003. The Spanish population has more than quadrupled, while the increase in the Portuguese population has been >1.5 (Chapron *et al.* 2014). However, in the last 10 years the population in Spain has remained mostly stable (Kaczensky *et al.* 2012). With respect to the geographic range of the species, it has experienced a 150% increase from 1950 to 2010 (Chapron *et al.* 2014).

Conclusions

The distribution of the wolf was more restricted and the abundance of this species was significantly lower in the past 60 years than nowadays. After a great increase from the 1950s until 2003 due to natural colonization, the population size in some areas has stabilized in the last 10 years. The magnitude of the overall increase in the Iberian population is about 400% in the indicated 50-year period, with some exceptions of population declines in Trás-os-Montes area and South Douro river area in Portugal.

Step 2.1 - FRP assessment

Given that the trend in population size has been positive or at least stable after a deep low, we note as well that wolf numbers in the Iberian Peninsula are still increasing. Apparently the wolf is still colonizing its natural range. We propose that the FRP for the NW Iberian population of the wolf should be in the order of magnitude of thousands of individuals (current population size=2000-2500 individuals). Instead of setting this number as the FRP, we use operators and suggest to constantly monitor the status of the population until range and population size are stable for a sufficient long time. This would be the final FRP value.

Step 2.2 - FRR assessment

Conclusions

For the NW Iberian wolf population (in ES and PT) FRP > CV.

Based on the previous discussion on the ongoing increase in the distribution of the population due to natural expansion since the 1950s, and following the stepwise approach, we conclude that the range can grow up to its natural range (i.e. likely to be higher than the current 120000 km^2) until monitoring points to stabilization for a sufficiently long time.

Conclusions

For the NW Iberian wolf population (in ES and PT) FRR > CV.

References

Boitani L., Lovari S., Vigna-Taglianti A. Fauna d'Italia. Mammalia III – Carnivora and Artiodactyla. Calderini, Bologna, Italy (2003).

Chapron, G., *et al.* Recovery of large carnivores in Europe's modern human-dominated landscapes. Science 346.6216: 1517-1519 (2014).

Kaczensky, P., Chapron, G., von Arx, M., Huber, D., Andrén, H., Linnell, J.D.C., Status, Management and Distribution of Large Carnivores – Bear, Lynx, Wolf & Wolverine – in Europe. European Commission, Brussels (2012).

Lucchini V., Galov A., Randi E. Evidence of genetic distinction and long- term population decline in wolves (Canis lupus) in the Italian Apennines. Mol Ecol 13(3):523–536 (2004).

Torres, R.T. & Fonseca, C. Perspectives on the Iberian wolf in Portugal: population trends and conservation threats. Biodiversity and Conservation 25: 411. doi:10.1007/s10531-016-1061-6 (2016).

Vilà C., Castroviejo J., Urios V. The Iberian wolf in Spain. In: Promberger C, Schröder W (eds) Wolves in Europe-status and perspectives. Munich Wildlife Society, Ettal, pp 104–109 (1993).

Vilà C., Wayne R.K. Hybridization between wolves and dogs. Conserv Biol 13(1): 195–198 (1999).

4 Birds

4.1 Smew (*Mergellus albellus*) in northern and western Europe

André van Kleunen, Marc van Roomen & Ruud Foppen

Step 1.1 - Biology of the species
Smew is a rather compact diving duck (38-44 cm, 0.5-0.9 kg). Male has a distinct white plumage with some black lines and patches. Females and juveniles are predominantly greyish coloured with a brownish head with white cheeks.

This species is highly migratory, often resting and feeding on inland waterbodies when on passage. It arrives on the breeding grounds from April or early-May and breeds from mid-May onwards in single pairs or loose groups. Males gather in large flocks close to the breeding grounds after mating to undergo a flightless moulting period. The species then leaves the breeding grounds in early-September and October. Outside of the breeding season the species is highly gregarious and occurs in small or large flocks usually not exceeding 100 individuals, although larger gatherings may form at major passage waters during migration and flocks of over 10,000 have been recorded during the winter. It breeds on freshwater oligotrophic lakes, pools, oxbow lakes, backwaters of large slow-flowing rivers, muskegs (bogs) and flooded riverside woods in the coniferous and mixed deciduous/evergreen forest zones. It shows a preference for shallow water up to 4 m (maximum 6 m) deep, and requires mature broadleaved trees (e.g. oaks, willows and aspens) with holes in which to nest. The species overwinters on large freshwater lakes and reservoirs, ice-free rivers, brackish coastal lagoons, estuaries and sheltered coastal bays (although rarely on the open sea), often resting and feeding on small bodies of water or small streams when on passage. The species feeds diurnally by diving to depths of 1-4 m and forms gregarious nocturnal roosts. During the breeding season its diet consists predominantly of benthic aquatic invertebrates such as adult and larval insects, crustaceans, molluscs and polychaete worms, as well as amphibians, fish and plant matter (seeds, leaves and roots). During the winter and in early spring however the species mainly feeds on fish. The species nests in tree hollows up to 10 m or more above the ground (especially those excavated by Black Woodpeckers *Dryocopus martius*) in mature broadleaved trees (e.g. oak, willow or aspen). It may also nest in artificial nest boxes, especially those erected to attract Common Goldeneye *Bucephala clangula* (source: BirdLife International 2017).

Step 1.2 - Spatial scale of functioning

Breeding population
The Smew's breeding distribution is large and extends from Sweden to eastern Siberia, mainly in boreal areas (del Hoyo *et al.* 1992). In the EU virtually the whole population occurs in central/northern Sweden and Finland. The EU27 breeding population of Smew is estimated at 3100-7600 pairs in 2008-2012 and its range is estimated at 179,000 km^2 and its distribution at 46,700 km^2 (European Environment Agency 2015).

The species' dispersal capacity is estimated at 250 km. Gaps larger than 250 km do not exist in the species' breeding distribution in the EU27 (Figure 4.1.1). So the North-European breeding population can be considered as one population (in fact continuing outside the EU27 boundaries in Russia) for which population category MR1 applies (widespread migratory species with more or less continuous distribution) and which FRVs are set at member state level.

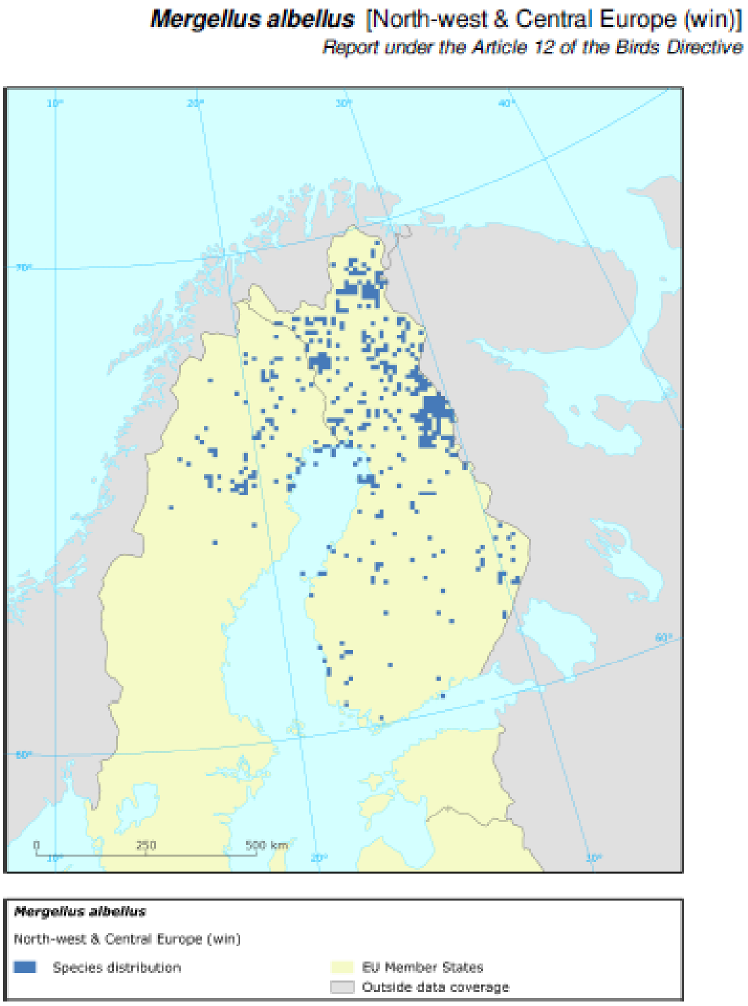


Figure 4.1.1 *Breeding distribution of Smew in EU27 in 2008 -2012 (10 km2-grid cells). Source: European Environment Agency (2015)*

Passage/wintering population
The North European breeding population is migratory. Wintering grounds are located in some wetland areas across the temperate and Mediterranean parts of Europe. In the flyway-approach three geographic populations are separated, two of them occurring in the EU27: one wintering in Southeast-Europe and the other wintering in Western Europe (Figure 4.1.2). Breeding birds from Sweden and Finland mainly migrate to the west and those from West-Russia to Southeast-Europe. The Northwest/Central-European flyway-population is estimated at 40,000 ex. and the Black Sea/East Mediterranean at 35,000 ex. (Wetlands International 2017). With regard to the EU27 member states: the Northwest/Central European wintering population held 23,444-37,920 ex. in 2008-2012 and the Southeast-European population 3356- 6780 ex (European Environment Agency 2015).

These populations correspond to category MNR3: species with one or a few isolated non-reproductive (wintering/staging) populations with FRVs to be set at supranational level.

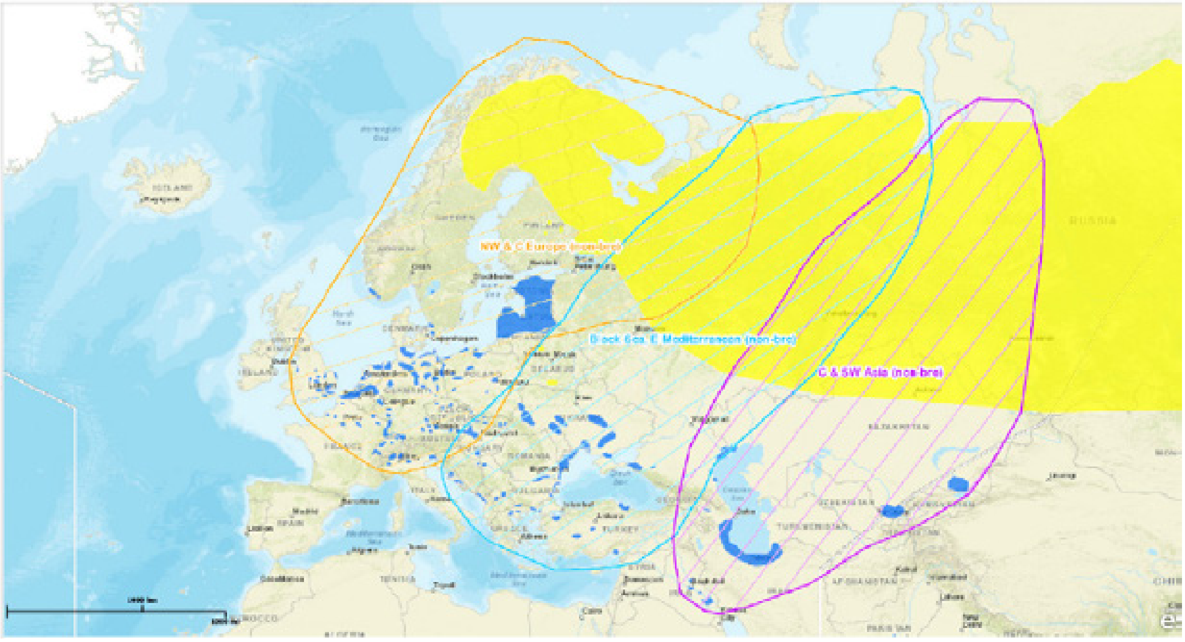


Figure 4.1.2 *Breeding areas (yellow), wintering areas (solid blue areas) and flyway-populations (open polygons) of Smew. Source: Wetlands International (2017)*

Conclusions

Smew's breeding distribution is large and extends from Sweden to eastern Siberia. The North-European breeding population can be considered as one population (continuing in Russia) corresponding to population category MR1: widespread migratory species with more or less continuous distribution for which FRVs must be set at member state level.

In the flyway-approach three separate geographic populations are distinguished, two of them occurring in the EU27: the Northwest/Central-European population and the Black Sea/East Mediterranean. Both correspond to population category MNR3: species with one or a few isolated non-reproductive (wintering/staging) populations for which FRVs must be set at supranational level.

Step 1.3 - Historical perspective: what happened to the species?

The North-European breeding population declined throughout the 19th and 20th centuries due to habitat degradation and loss (e.g. the loss of mature trees in river valleys as a result of logging, conversion to agriculture and river canalisation). The species has also suffered local declines as a result of predation by American mink *Neovison vison* (BirdLife International 2017). However since 1980 both the Swedish and Finnish breeding populations have increased in numbers (European Environment Agency 2015).

Since the 1990s wintering numbers of the West/Central European population have been stable, but probably numbers of the Southeast European wintering population have declined (Wetlands International 2017), possibly due to hunting pressure during migration. The species is also suspected to have altered its wintering range because of climate change.

Step 1.4 - Analysis of distribution and trends

Breeding population

Distribution data and data on numbers are available from Article 12 Bird Directive reporting 2008-2012 (European Environment Agency 2015) and Hagemeijer & Blair (1997). These data don't show negative trends in breeding population size at Member State level since 1980. It is not possible to quantify the species' numbers and distribution further back in time. There are indications that historically, back to the 19th century, numbers were higher, as was its distribution. In these times the species occurred in the eastern Europe where it is almost absent nowadays.

Wintering population

For most European countries data on wintering numbers and distribution are collected at site level in the International Waterbird winter census (Wetlands International) since the 1970s. Furthermore trends and population estimates were published in the Bird Directive reporting 2008-2012 (European Environment Agency 2015). There are no negative trends in the Northwest- and Central-European wintering population since the 1980s. The East-European wintering population in the Black Sea and East Mediterranean possibly has declined between 1990-2012.

Conclusions

The smew's breeding distribution doesn't show a negative trend since 1980 but historical numbers and distribution were probably larger than nowadays.

The Northwest- and Central-European wintering population is stable since the 1980s. The East-European wintering population possibly has declined between 1990-2012 – extent unknown.

Step 2.1 - FRP assessment

Breeding population

A species specific PVA is not available. The upscaled MVP is estimated at 12,500 pairs. So this is considered the bottom line for the Smew population. Although the EU27 breeding population does not exceed this value (3100-7600 pairs) the whole European population (the population continues in Russia) probably does (with 9,200-17,600 pairs). This population does even further extend into Asia. Regarding EU27, current numbers exceed the population size in 1980, given the increase in numbers since then in Sweden and Finland. In historical perspective this can be considered as a population recovery. Because current numbers clearly exceed MVP and there are no indications of a recent decline in distribution (see step 2.2 FRR assessment) we suggest to take current values as FRP's, reflecting a relatively favourable situation: Sweden: 1100-2000 pairs, Finland: 2000-5500 pairs (European Environment Agency 2015).

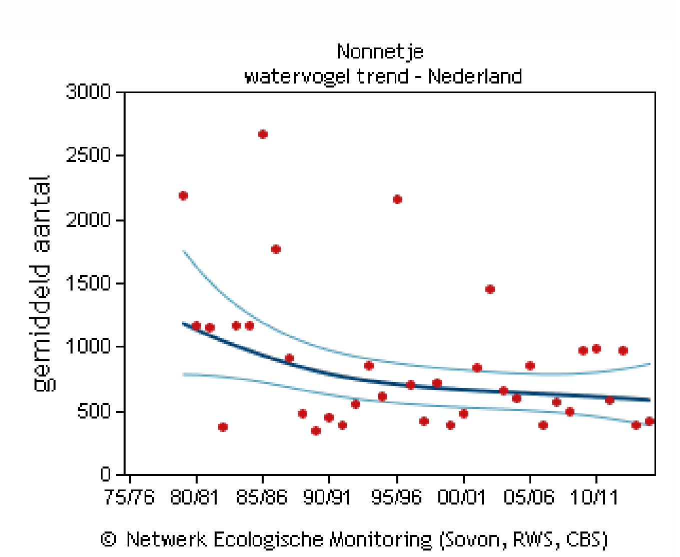


Figure 4.1.3 *Trend of Smew in the Netherlands based on annual seasonal average numbers (Source Sovon)*

Wintering population in the North-west European flyway

Since the trend for the North-west European flyway population is positive since 1980, probably a recovery of declines in the past, current value can be used for the supranational FRV: 26,800-44,700 individuals (European Environment Agency 2015).

National targets should be based on the carrying capacity of sites/habitat, as the species shows spatial dynamics in winter steered by climatic conditions. E.g. in harsh winters birds wintering in the Baltic Sea move to western Europe. Peak numbers observed over several years can be used as a proxy for setting targets. In the Netherlands the wintering numbers of Smew have declined since 1980. This can be explained by a northerly shift in wintering areas. We propose to use peak-year numbers to set the national target for the Netherlands: 2500 ex. expressed as seasonal average numbers (Figure 4.1.3).

Conclusions

Breeding population: FRP's for Smew in Sweden and Finland can be based on current values reflecting long-term positive trends following historical decline:
FRP = CV (SE 1100-2000 pairs; FI 2000-5500 pairs).

Wintering population: the supranational FRP for the north-west European flyway population corresponds to current value:
FRP = CV (26,800-44,700 individuals, supranational).

National targets for wintering must consider the carrying capacity of the member state for which peak numbers can be used as proxy.

Step 2.2 - FRR assessment

Breeding population
There are no indications that the species' distribution in Northern Europe has declined or changed in configuration in the last decades. Current values are used to set FRR's: Sweden: 9000 km^2, Finland: 37700 km^2 (European Environment Agency 2015).

Wintering population
Accumulated distribution over several years, derived from maps including the distribution of wintering populations in peak years can be used as 'current value' for the supranational FRR.

Conclusions

Breeding population: FRR's for Sweden and Finland can be based on current values reflecting long-term positive trends:
FRR = CV (SE 9000 km^2, FI 37,700 km^2).

Wintering population: the supranational FRR for the north-west European flyway population corresponds to current value to be derived from maps over several years, including peak years.

References

BirdLife International (2017) Species factsheet: Mergellus albellus. Downloaded from http://www.birdlife.org on 08/03/2017.
European Environment Agency 2015: https://bd.eionet.europa.eu/article12/summary?period=1&subject=A332
Del Hoyo, J.; Elliot, A.; Sargatal, J. 1992. Handbook of the Birds of the World, vol. 1: Ostrich to Ducks.
Wetlands International (2017). "Waterbird Population Estimates". Retrieved from wpe.wetlands.org on Thursday 16 Mar 2017
Hagemeijer E.J.M. & Blaie M.J. (eds.) 1997. The EBCC Atlas of the European Breeding Birds: Their Distribution and Abundance. T. & A. D. Poyser, Lon don.Hilbers J.P., Schipper A. M., Hendriks A. J., Verones F., Pereira H. M.,& Huijbregts M. A. J. 2016. An allometric approach to quantify the extinction vulnerability of birds and mammals Ecology 97(3): 615-626.

4.2 Spotted Crake (*Porzana porzana*) in the EU with focus on the Netherlands

André van Kleunen, Marc van Roomen & Ruud Foppen

Step 1.1 - Biology of the species

Spotted Crake is a small (l 19-22,5 cm, 0.09 kg) rail with a greenish-brown plumage with dark streaks and some spots across its body. Its neck is greyish and it has distinct whitish under tail. Its legs are greenish with long toes. Its bill is greenish-yellow with reddish base. The species is rather elusive and is more often heard than seen; it has a distinctive call performed in spring mostly at night.

This species is fully migratory. Its autumn dispersal to its wintering grounds beginning mid-July, with the species returning to reoccupy its breeding grounds from April. The species breeds in Europe from April to July and in the former USSR from May to July. Early migrating birds (mainly juveniles) often moult in August during stops on migration, during which they become flightless for c.3 weeks. The species is territorial throughout both breeding and non-breeding seasons and is usually seen singly, in pairs or in family groups, although occasionally small groups of 2-4 individuals may forage together on migration. It normally roosts at night in thick vegetation and forages by day (although this behaviour is reversed when migrating). The species inhabits similar habitats in both its breeding and winter ranges, and generally requires very shallow water (less than c.15 cm deep, typically foraging in water less than 7 cm deep) that is rich in invertebrate food and is interspersed with stands of low vegetation cover. The species shows a preference for freshwater wetlands with a range of water depths or where water levels vary seasonally, especially where these have a mixture of muddy, moist and shallowly flooded substrates and a dense covering of grass, sedges, rushes, Polygonum, Iris, Equisetum and other emergents, as well as trees (e.g. Acacia, Sesbania, Betula, Salix and Alnus). Suitable habitats include seasonal and permanent marshes and fens, bogs, damp meadows, the edges of drainage ditches, swamps, seasonally flooded pans, pools in flooded grassland, grassy margins of reservoirs and lakes, slow-flowing rivers and sewage settling-ponds. The species is omnivorous, its diet consisting of small aquatic insect adults and larvae (e.g. Trichoptera, Odonata, Diptera, Coleoptera, Hemiptera, Lepidoptera and ants), earthworms, molluscs, arachnids (e.g. spiders and water mites) and small stranded fish (1-2 cm long), as well as algae and the shoots, leaves, roots and seeds of *Panicum*, *Oryza*, *Carex* and Schoenoplectus. The nest is a thick-walled cup of plant matter, usually placed in thick vegetation near or over standing water, or alternatively in a tussock, or built up well above the water level (source: BirdLife International 2017).

Step 1.2 - Spatial scale of functioning

Breeding population

The species' global range covers large parts of Europe, north up to Southern-Scandinavia and extends east to Central-Asia (Figure 4.2.1). East-European countries harbour largest numbers and have the highest degree of occupancy. In southern Europe and western Europe the species is more localised and numbers can show quite strong year to year variations as a consequence of habitat conditions (BirdLife International 2017). For instance summer floodings of the river Rhine floodplain in the Netherlands could lead to a tenfold increase in numbers of calling males (Vogel & van der Wal 1988, Van den Bergh & Helmer 1984).

The EU27 breeding populations is estimated at 9900-26,700 males in 2008-2012 (European Environment Agency 2015) and its distribution at 234,300 km^2.
The species' dispersal capacity is estimated at 50 km. Such gaps do not exist in the species' distribution in Europe, apart from the British Isles, parts of Italy and Spain. However the species irruptive behaviour might suggest a higher dispersal capacity than estimated with the allometric approach used. Therefore we propose to consider the Spotted Crake at EU27 scale as a widespread migratory species (population category MR1).

Figure 4.2.1 *Breeding distribution of Spotted Crake in EU27 in 2008 -2012 (10 km2-grid cells). Source: European Environment Agency (2015)*

Passage population
The Spotted Crake is migratory and its wintering grounds are located in Africa and in southern Central-Asia. Comparatively little is known about its abundance and distribution during migration in Europa, but it is known to turn up in the same habitat where the species does breed. So it is likely to match with its breeding range in Europe. Therefore, and because no quantitative assessment can be made concerning the number of migrating individuals, we propose to set FRVs for the breeding population only.

Conclusions

Spotted Crake at EU27 scale is a widespread migratory species with a more or less continuous breeding distribution (population category MR1) for which FRVs must be set at the national level.

We propose to set FRVs for the breeding population only because during migration the distribution of Spotted crake is likely to match with its breeding range in Europe.

Step 1.3 - Historical perspective: what happened to the species?
Since 1980 numbers in the EU27-countries together have been stable (European Environment Agency 2015). Further back in the 20th century numbers and distribution of Spotted Crake are believed to have declined as a consequence of wetland drainage or hydrological changes, changes in water quality and agricultural intensification, in particular in Western-Europe (BirdLife International 2017). The magnitude of the decline cannot be quantified as historical numbers are not available. Little is known about the population status of the passage population.

Step 1.4 - Analysis of distribution and trends (breeding population)

Distribution data and data on numbers are available from Article 12 Bird Directive reporting 2008-2012 (European Environment Agency 2015) and Hagemeijer & Blair (1997). It is likely that current numbers and the distribution in the EU are lower than in the early 20th century. Since 1980 numbers at EU27 level have been stable. However at member state level there is some variation, e.g. declines in Finland (31-88%), Italy (magnitude unknown), the Netherlands(18-34%) and Slovakia (10-24%).

Conclusions

The Spotted Crake's breeding distribution and numbers don't show a negative trend since 1980 but historical numbers and distribution were probably larger than nowadays.

Step 2.1 - FRP assessment (breeding population)

A species specific PVA is not available. The upscaled MVP is estimated at 12,500 breeding pairs. The current EU27-population (9900-26,700 calling males) exceeds this value and even more if we take into account the numbers in non EU-countries that are likely to belong to the same population. The species has been stable EU-wide both in numbers and distribution since 1980. Historical numbers might have been higher, but are not available. This species is classified in FRV-category MR1. This means that FRP's should be set at member state level, at least at 1980-size.

Although the Dutch population numbers show strong year-to-year fluctuations the average numbers in recent years are significantly lower than around 1980 (18-35% decline). So we propose to set the 1980 level as FRP. According to the Dutch Breeding bird atlas (Sovon Vogelonderzoek 2002) the breeding population of Spotted Crake in the Netherlands in 1980 was estimated at 150-1000 pairs, which is the (minimum value for the) FRP.

Conclusions

Breeding population: In the Netherlands the species has declined since 1980 (150-1000 pairs then), so FRP > CV (150-1000 pairs).

Step 2.2 - FRR assessment (breeding population)

We propose to set the reference periods for FRR's as is reasoned for setting FRP's above. In the Netherlands the species' distribution shows strong year-to-year fluctuations but on average has not changed since 1980 (SOVON 1987, SOVON (2017). The species' distribution is likely to have been larger earlier in the 20th century, but we don't have historical numbers. Based on the number of occupied 5x5 km squares the distribution for 1980 (SOVON 1987) is calculated at 3550 km2., which is the minimum value for the FRR.

Conclusions

The national FRR for Spotted Crake in the Netherlands is based on the distribution of the species around 1980: 3550 km^2. Earlier in the 20th century distribution was probably larger, so, based on the 1980-reference, FRR > 3550 km^2.

References

Van den Bergh L.M.J. & Helmer J. 1984. Over het voorkomen van Porseleinhoentjes Porzana porzana langs de Grote Rivieren in 1983. Vogeljaar 32: 279-289.

BirdLife International 2017. Species factsheet: Porzana porzana. Downloaded from http://www.birdlife.org on 08/03/2017.

European Environment Agency 2015:
https://bd.eionet.europa.eu/article12/summary?period=1&subject=A332

SOVON 1987. Atlas van de Nederlandse Vogels.

Vogel R.L. & van der Wal R. 1988. Het Porseleinhoentje Porzana porzana langs de IJssel in 1987. Limosa 61: 45-46.

4.3 Great Bustard (*Otis tarda*) in the EU with focus on the Iberian peninsula

André van Kleunen, Marc van Roomen & Ruud Foppen

Step 1.1 - Biology of the species
The Great Bustard is a large and heavy ground bird (male 6-18 kg, 105 cm). It has whitish underparts, greyish neck and brownish upperparts with black bars. The male has a rusty orange breast and tail and conspicuous skin-barbs in breeding plumage.

Originally it is a species of the Eurasian steppe, but has acclimated to agricultural landscapes. It occurs in open, flat or somewhat rolling landscapes, usually with a mixture of crops (cereals, vineyards, fodder plants), in some countries also with steppic grassland. Areas with little or no disturbance and abundant supply of insects are required for successful breeding. Nest sites are selected in grassland, fallow or cereal fields (primarily alfalfa in Central Europe and wheat in Russia, Mongolia and Kazakhstan) in areas of low patch-type diversity, far from human infrastructure and with good horizontal visibility. Highly variable migratory behaviour across populations, including obligate winter migrants (Asia, Russia), facultative migrants (central European populations) and partial winter and summer migrants with differential migratory pattern by sex (Iberian populations).

Its diet consist of plants, including composites, legumes, crucifers and grasses, but also invertebrates: Orthoptera and Coleoptera. A high proportion of cultivated plants is utilized from late summer through to winter and insects from late spring into late summer. Nesting season lasts from april-june. The simple nest is located on the ground. First breeding takes place after 5-6 years for males and 2-3 years for females (sources: BirdLife International 2017, Del Hoyo *et al.* 1996)

Step 1.2 - Spatial scale of functioning

Breeding population
Its current breeding range in Europa is highly fragmented with breeding populations occurring in the Iberian peninsula, Eastern Germany, Austria-Hungary, Romania, Ukraine, Belarus and Russia (Figure 1). Outside Europe its range extends in Central-Asia east to Mongolia and in Northern Africa in Marroc. The populations in Eastern Russia and Mongolia are considered a separate subspecies (*Otis tarda dybowskii*). The EU27 breeding population of the Great Bustard is estimated at 15,100-18,100 males in 2008-2012. Its EU27 distribution covers 82,000 km^2 (European Environment Agency 2015). The species' dispersal capacity is estimated at 50 km. Applying this distance to its (contracted and fragmented!) distribution we should consider the following metapopulations, in accordance with the geographical populations distinguished by Nagy (2009):

Fully EU:
- Iberian Peninsula
- Eastern Germany/Poland (German-Polish plain)

Partially EU:
- Central-Europe – Carpathian Basin(Austria/Hungary – historically also Czech Republic, Slovakia, Serbia, Romania and Bulgaria)

Europe –non EU:
- European Russia, Ukrain, Moldova.

Passage/wintering population
Most populations of the western-subspecies (*Otis tarda tarda*), occurring in Europe are considered resident, although some central-European populations and even the Iberian show some (short-distance) displacements. Particularly in harsh winters the species can show movements to areas well outside its normal range. We propose to consider the species sedentary with regard to FRV-setting.

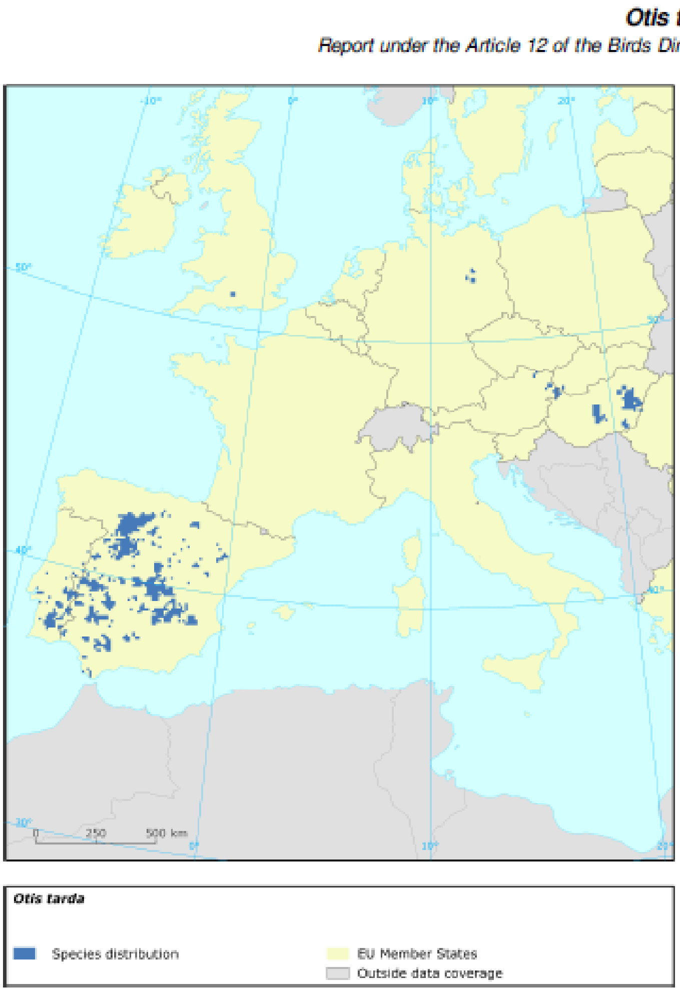


Figure 4.3.1 *Breeding distribution Great Bustard in EU27 in 2008 -2012 (10 km^2-grid cells). Source: European Environment Agency (2015)*

Conclusions

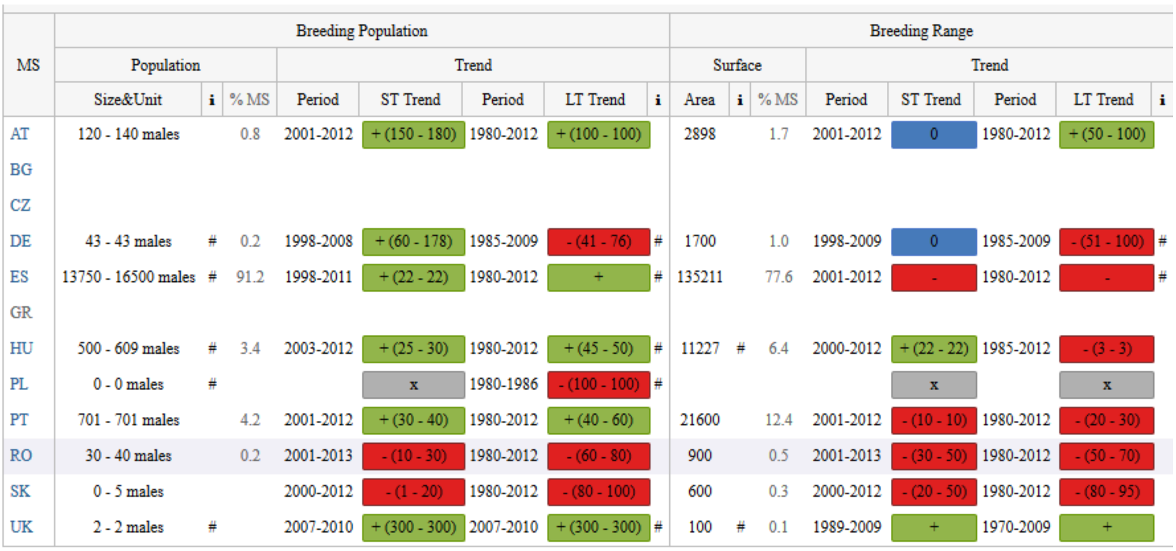
Breeding populations of the resident Great Bustard correspond to population category S3: uncommon species with small home ranges and one or a few populations at supranational level. For this category FRVs must be set at the supranational level.

Step 1.3 - Historical perspective: what happened to the species?

Historically the Great Bustard probably reached its maximum extension during the 18th century, when it benefited from extensive forest clearance. Subsequently it began to retreat due to habitat changes caused by human population growth, intensification of farming practices and hunting (Hagemeijer & Blair 1997). Its range has contracted and populations have become isolated. The EU 27 trend is positive since 1980, mainly due to the increase of the Iberian population as a consequence of successfully implemented conservation measures (BirdLife International 2017). Despite some recent recovery in some East-European countries current numbers are still lower than in 1980. The species continues to decline in Romania and Slovakia.

Regarding the species' range the EU27 picture since 1980 is even more negative: apart from Austria the species' range has contracted further, even in the Iberian peninsula.

Table 4.3.1 *Overview of population numbers, ranges and changes in this in EU-member states as reported for the Bird Directive reporting 2008-2012 (European Environment Agency 2015)*



MS	Breeding Population								Breeding Range							
	Population			Trend					Surface			Trend				
	Size&Unit	i	% MS	Period	ST Trend	Period	LT Trend	i	Area	i	% MS	Period	ST Trend	Period	LT Trend	i
AT	120 - 140 males		0.8	2001-2012	+ (150 - 180)	1980-2012	+ (100 - 100)		2898		1.7	2001-2012	0	1980-2012	+ (50 - 100)	
BG																
CZ																
DE	43 - 43 males	#	0.2	1998-2008	+ (60 - 178)	1985-2009	- (41 - 76)	#	1700		1.0	1998-2009	0	1985-2009	- (51 - 100)	#
ES	13750 - 16500 males	#	91.2	1998-2011	+ (22 - 22)	1980-2012	+	#	135211		77.6	2001-2012	-	1980-2012	-	#
GR																
HU	500 - 609 males	#	3.4	2003-2012	+ (25 - 30)	1980-2012	+ (45 - 50)	#	11227	#	6.4	2000-2012	+ (22 - 22)	1985-2012	- (3 - 3)	
PL	0 - 0 males	#			x	1980-1986	- (100 - 100)	#					x		x	
PT	701 - 701 males		4.2	2001-2012	+ (30 - 40)	1980-2012	+ (40 - 60)		21600		12.4	2001-2012	- (10 - 10)	1980-2012	- (20 - 30)	
RO	30 - 40 males		0.2	2001-2013	- (10 - 30)	1980-2012	- (60 - 80)		900		0.5	2001-2013	- (30 - 50)	1980-2012	- (50 - 70)	
SK	0 - 5 males			2000-2012	- (1 - 20)	1980-2012	- (80 - 100)		600		0.3	2000-2012	- (20 - 50)	1980-2012	- (80 - 95)	
UK	2 - 2 males	#		2007-2010	+ (300 - 300)	2007-2010	+ (300 - 300)	#	100	#	0.1	1989-2009	+	1970-2009	+	

Step 1.4 - Analysis of distribution and trends

Distribution data and data on numbers are available from Article 12 Bird Directive reporting 2008-2012 (Table 4.3.1; European Environment Agency 2015) and Hagemeijer & Blair (1997). Apart from these generic sources at least a part of the European populations has been studied relatively well and an overview was compiled in the species' action plan (Nagy 2009).

Historically numbers and distribution have declined before 1980 (see step 1.3). Since 1980 numbers have increased at EU27 level, but its distribution trend remained negative.

Conclusions

The Great Bustard's numbers EU-wide have recovered since 1980, but its distribution has decreased.

Step 2.1 - FRP assessment

A species specific PVA is not available. The upscaled MVP is estimated at 2500 males.

The Iberian population has declined strongly in historical time (magnitude not available). The 1980 population encompassed about 7.000 males. Since 1980 it has recovered and in 2008-2012 it counted 14,000-17,000 males. The upscaled MVP value has already been exceeded since 1980. As the 1980 population was depleted as a consequence of anthropogenic pressures (hunting and changes in farming practices) we don't consider the 1980-value a proper value for the FRP. Recently the numbers have recovered strongly as consequence of protection measures (in particular by a ban on hunting). This is likely to be a better approach of the FRP of this population, at least the minimum value and even so given the decline in distribution. We propose FRP ≥ CV (14,000-17,000 males).

The population in the German-Polish plain harboured only 43 males in 2008-2012. This remnant population is depleted as a consequence of hunting and in particular habitat deterioration (farming practices). In the seventies it comprised 800 males (Blair & Hagemeijer 1997). It is likely that further back in time numbers were higher (magnitude unknown). The upscaled MVP (2500 males) was even not reached In the seventies. FRP should be higher than 2500 males.

Conclusions

Iberian Peninsula: Since 1980 population numbers of Great Bustard have at least partly recovered from depletion to 14,000-17,000 males. Further recovery might be possible. FRP > 14,000-17,000 males.

German-Polish plain: Population is depleted and far below the upscaled MVP even in the seventies. FRP >> 2500 males.

Step 2.2 - FRR assessment

The range and distribution of the Iberian population have declined since 1980, although the magnitude is unknown (Table 1). Although current population numbers are higher than in 1980, its distribution has not recovered at all. We propose to use the 1980 distribution to set the FRR. In the first European Breeding Bird Atlas the species' distribution covered at least 202,500 km^2 squares (Hagemeijer & Blair 1997). This might even be a minimum as the distribution most likely declined before 1980 already because of hunting and adverse farming practices.

The range of the German-Polish population in Germany has decreased by 50-100% since 1980, only 900 km^2 (10x10 km squares) is left. The species has disappeared from Poland in 1986 where it occurred in the 1960's up to Central-Poland (Clutz von Blotzheim *et al.* 1994). Based on Blair& Hagemeijer its seems likely that the historical distribution before the seventies covered at least 10 50x50 km squares in Eastern-Germany only. A similar figure seems likely for Poland. So we propose a minimum FRR of 50,000 km^2 (based on occupied 50x50 km squares).

Conclusions

Iberia: FRR should equal or exceed the 1980 value. FRR ≥ 202,500 km^2.

German-Polish plain: FRR ≥ 50,000 km2 (50x50 km square level).

References

BirdLife International (2017) Species factsheet: Otis tarda. Downloaded from http://www.birdlife.org on 08/03/2017.

Clutz von Blotzheim U.N., Bauer K.M. & Bezzel E. 1989. Handbuch der Vögel Mitteleuropas. Band 5 Galliformes und Gruiformes. Aula Verlag, Wiesbaden.

European Environment Agency 2015: https://bd.eionet.europa.eu/article12/summary?period=1&subject=A332

Hagemeijer E.J.M. & Blair M.J. (eds.) 1997. The EBCC Atlas of the European Breeding Birds: Their Distribution and Abundance. T. & A. D. Poyser, London.

Del Hoyo J., Elliott A. & Sargatal J. (eds.) 1996. Handbook of the Birds of the World. Vol. 3. Hoatzins to Auks. Lynx Edicions, Barcelona.

Nagy S. 2009. International single species action plan for the Western Palearctic population of Great Bustard, Otis tarda tarda.

4.4 Dunlin (*Calidris alpina*) in western Europe and the Baltic area

André van Kleunen, Marc van Roomen & Ruud Foppen

Step 1.1 - Biology of the species
Rather small wader species (l: 17-21 cm, 33-85 g), has slightly decurved bill. In summer plumage reddish-brown upperparts, white underparts with distinctive black patch on belly. In winter plumage uniform greyish and white underparts.

This species is a fully migratory circumpolar breeder with several sub-populations that employ a number of migration strategies, from short coastal flights to long, non-stop flights overland on a broad front. The sub-population that breeds in north-east Greenland migrates through Iceland, Britain and western France to arrive in its West African wintering grounds (specifically Banc d'Arguin in Mauritania) from late-July, returning again between March and early-April. European birds may gather in large congregations from the beginning of July in areas such as the Wadden Sea or the Wash to moult, and some juveniles may remain in the non-breeding range all year. The species breeds dispersed or aggregated in loose colonies, and travels in group sizes of up to 1,500 on passage, remaining in large groups (up to hundreds of thousands of birds) throughout the non-breeding season. The species is active both diurnally and nocturnally. Habitat Breeding In the breeding season this species frequents moist boggy ground interspersed with surface water, such as tussock tundra and peat-hummock tundra in the arctic, as well as wet coastal grasslands, salt marshes and wet upland moorland. Non-breeding In the non-breeding season this species mainly prefer estuarine mudflats, but also frequent a wide variety of freshwater and brackish wetlands, both coastal and inland, including lagoons, muddy freshwater shores, tidal rivers, flooded fields, sewage farms, salt-works, sandy coasts, lakes and dams. For roosting during high tides and at night this species prefers large fields of naturally fertilised short pasture or soil-based crops with few vertical structures that could be used by predators. Diet Breeding This species is omnivorous during the breeding season, consuming mostly adult and larval insects (dipteran flies, beetles, caddisflies, wasps, sawflies and mayflies), and also spiders, mites, earthworms, snails, slugs and plant matter (usually seeds). Non-breeding It is also omnivorous during the non-breeding season, consuming mostly polychaete worms and small gastropods, as well as insects (dipteran flies and beetles), crustaceans, bivalves, plant matter and occasionally small fish. Breeding site Its nest is a scrape or shallow depression in the ground, concealed in vegetation and sometimes in a tuft or tussock (and thus raised slightly off the ground). Management information The provision of well-surfaced paths in breeding areas that receive > 30 visitors a day has been shown to reduce the impact of human disturbance on this species' reproductive performance. It is also known to show increased hatching successes when ground predators have been excluded by erecting protective fences around nesting areas (source: BirdLife International 2017).

Step 1.2 - Spatial scale of functioning

Reproductive population
The Dunlin has a circumpolar breeding range. In Europe it occurs south to northern Poland, Germany and Southwest England. Four breeding (flyway) populations are distinguished in Europe (Delany *et al.* 2009; Figure 4.4.1, Table 4.4.1):

1. *C. a. alpina* breeding in northern Fennoscandia and Siberian Russia
2. *C. a. schinzii* from Iceland
3. *C. a. schinzii* from Ireland, Britain and Southwest Norway
4. *C. a. schinzii* from the Baltic countries.

The total European breeding population is estimated at 426,000-562,000 pairs (BirdLife International 2017). The overall population trend is negative, although some populations are stable or have unknown trends (Wetlands International 2015, Table 2).

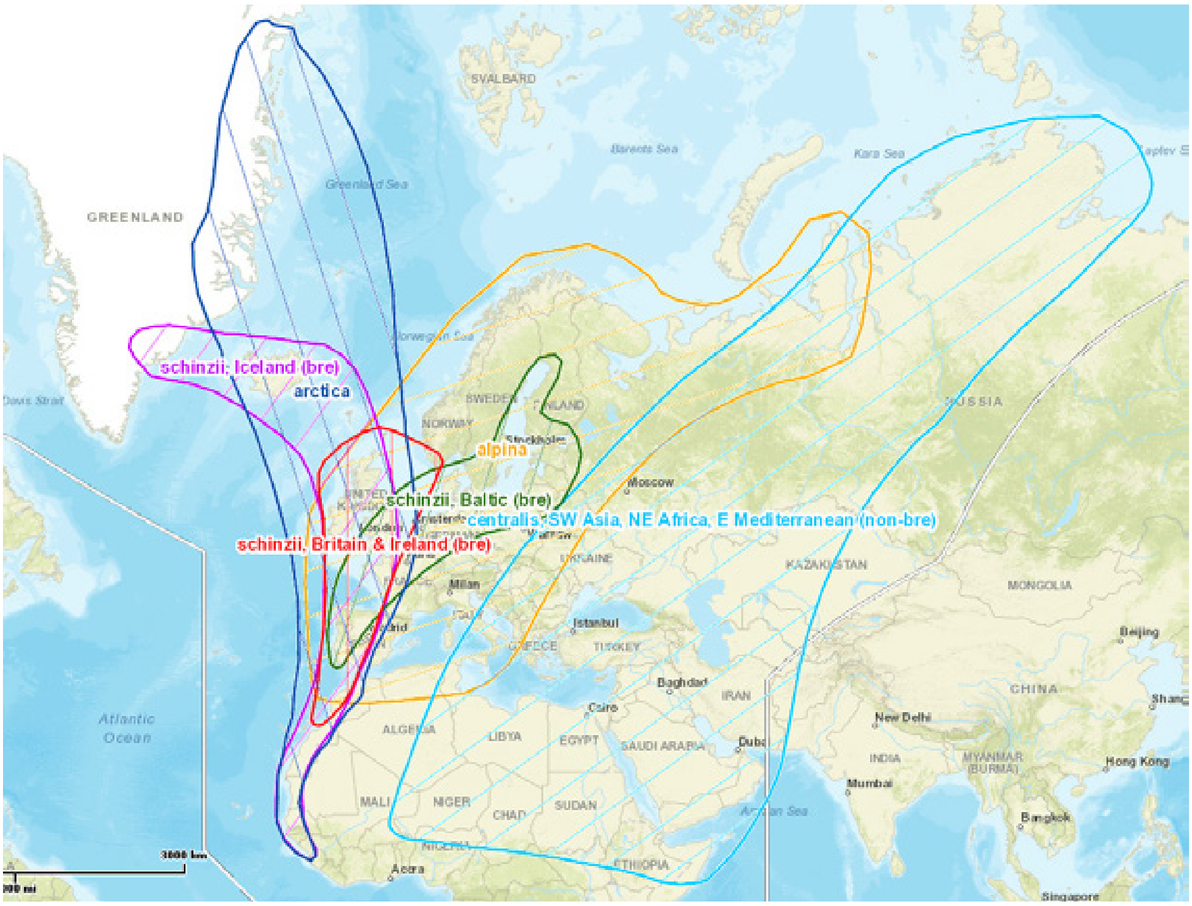


Figure 4.4.1 *Flyway-populations of Dunlin, including breeding, passage and wintering range. Source: Wetlands International (2017)*

Table 4.4.1 *Flyway populations of Dunlin occurring on passage and/or wintering in Europe (Delany et al 2009)*

Population Name	Subspecies	Breeding range	Nonbreeding range
North Scandinavia-West Siberia and West Europe	*C. a. alpina*	North Scandinavia & West Siberia	West Europe, west Mediterranean
Iceland & Greenland/NW and West Africa	*C.a. schinzii*	Iceland	West Africa
Baltic/SW Europe & NW Africa	*C.a. schinzii*	Baltic	SW Europe & NW Africa
Britain & Ireland/SW Europe & NW Africa	*C.a. schinzii*	Britain, Ireland, S Norway	SW Europe & NW Africa
NE Greenland/West Africa	*C.a. arctica*	NE Greenland	West Africa?
South-west Asia, North-east Africa & East Mediterranean	*C.a. centralis*	North Central Siberia	East Europe, East Mediterranean, Middle East

We propose to treat the flyway breeding populations of Dunlin as categories MR3: C.a. *schinzii* in Iceland, C. a. *schinzii* in Ireland, UK and Southwest Norway, C.a. *schinzii* in Wadden Sea and Baltic countries and C. a. *alpina* in North Scandinavia and West Russia). Category MR3 corresponds to migratory species with individuals with small home ranges and one or a few supranational populations.

Passage/wintering population

All Dunlin populations are migratory and show directional movements to wintering areas. In Europe these winter areas encompass coastal areas in Western Europe in the Atlantic region and in Southern Europe at the Mediterranean coast. Six flyway populations are distinguished that winter or occur on passage in Europe (Table 4.4.1).

During passage and to a lesser extent wintering considerable mixing of the different flyway populations takes place. Furthermore the different populations are not separable in the field. Some differences in timing of migration between the populations occurs, especially in spring. We propose to treat the different passage/wintering populations of Dunlin for setting FRVs as:

- Passage and wintering Dunlin in Western Europe and North Africa. This is predominantly *alpina* in winter with small numbers of *schinzii* from Ireland/UK/South Norway and *schinzii* from Baltic countries. During migration extra *schinzii* from Iceland and *arctica* occur in France, Ireland and UK.
- Passage and wintering Dunlin in Eastern Europe (from South Italy east) and Eastern Mediterranean. This is predominantly *centralis*.
- (Not threated within EU BD scope): Wintering Dunlin in West Africa. This is predominantly *schinzii* from Iceland with *arctica* from East Greenland.

All these populations correspond to category MNR3 (migratory species with one or a few isolated non-reproductive (wintering/staging) populations at supranational level) with FRVs to be set at supranational level.

Conclusions

All breeding populations of Dunlin correspond to FRV category MR3. Passage and wintering populations of Dunlin belong to FRV category MNR3.

Step 1.3 - Historical perspective: what happened to the species?

The overall population trend is decreasing, although some populations are stable or have unknown trends (Wetlands International 2015, Table 2). The population of *C. a. schinzii* breeding in Iceland seems to be stable. However numbers and probably also distribution of the Baltic population in particular have strongly declined as a consequence of afforestation, agricultural intensification and the decrease or cessation of coastal grazing. During passage and in the winter Dunlin is restricted to a small number of estuaries, so is vulnerable to any changes in quantity and quality of this habitat, for example through land reclamation (Hagemeijer & Blair 1997, BirdLife International 2017).

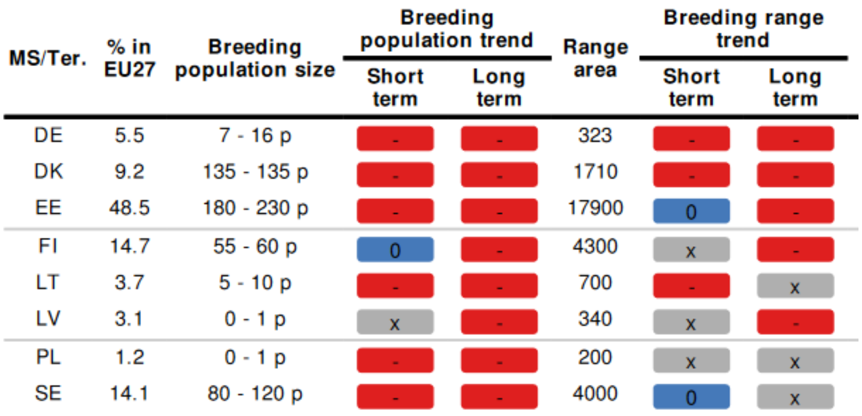
Step 1.4 - Analysis of distribution and trends

Breeding population

Distribution data and data on numbers are available from Article 12 Bird Directive reporting 2008-2012 (European Environment Agency 2015) and Hagemeijer & Blair (1997). The breeding numbers of *C. a. alpina* in Finland have decreased since 1980 (40-50%). However in Sweden in the same period breeding numbers of *C. a. alpina* have increased (10-30%). The numbers of *C. a. schinzii* in the British Isles are stable or have decreased slightly; large population UK (8600-10600) stable, small population Ireland (150 pairs) 58-79% decrease. Its range has contracted slightly. The breeding numbers of *C. a. schinzii* on Iceland are considered to be stable (Delany *et al.* 2009).

The numbers of the Baltic Sea-area population of *C. a. schinzii* have decreased since 1980 at least 50%, but probably more (Table 4.4.2). Although difficult to quantify a strong decline already took place before 1980. E.g. in the past densities up to 50 pairs/km^2 were recorded. Nowadays the total breeding population is estimated at around 500 breeding pairs and is considered depleted. The range has decreased as well, probably more than 30% since 1980 (European Environment Agency 2015). The species has even disappeared as regular breeding bird in the southern Wadden Sea, for instance.

Table 4.4.2 *Population estimates and trends in numbers and range of the Baltic breeding population of C. a. schinzii in 2008-2012 at EU-member state level (European Environment Agency 2015)*



MS/Ter.	% in EU27	Breeding population size	Breeding population trend		Range area	Breeding range trend	
			Short term	Long term		Short term	Long term
DE	5.5	7 - 16 p	-	-	323	-	-
DK	9.2	135 - 135 p	-	-	1710	-	-
EE	48.5	180 - 230 p	-	-	17900	0	-
FI	14.7	55 - 60 p	0	-	4300	x	-
LT	3.7	5 - 10 p	-	-	700	-	x
LV	3.1	0 - 1 p	x	-	340	x	-
PL	1.2	0 - 1 p	-	-	200	x	x
SE	14.1	80 - 120 p	-	-	4000	0	x

Passage/wintering population
Since the 1970s for most European countries data on wintering numbers and distribution are collected at site level in the International Waterbird winter census (Wetlands International 2017). Furthermore trends and population estimates were published in the Bird Directive reporting 2008-2012 (European Environment Agency 2015). Table 4.4.3 summarizes the flyway-population trends. *C. alpina* and *C. a. schinzii* breeding in the Baltics have declined, *C.a schinzii* breeding in Iceland is stable/fluctuating, *C.a. schinzii* breeding in the British isles has increased, at least between 1996-2012. However the Bird Directive data suggest a slight decrease since 1980.

Table 4.4.3 *Numbers and trends of flyway-populations of Dunlin (source: Wetlands International)*

Population	Start Year Size	End Year Size	Minimum	Maximum	Start Year Trend	End Year Trend	Trend
alpina, NE Europe & NW Siberia/W Europe & NW Africa	2000	2012	1.330.000	1.330.000	2003	2014	declining
schinzii, Iceland & Greenland/NW and West Africa	2010	2014	730.000	830.000	2003	2014	stable/ fluctuating
schinzii, Baltic/SW Europe & NW Africa	1990	2012	1.180	1.430	2000	2012	declining
schinzii, Britain & Ireland/SW Europe & NW Africa	2005	2008	26.300	32.300	1996	2010	increasing
arctica, NE Greenland/West Africa	1996	1999	21.000	45.000	1988	2000	stable?

Step 2.1 - FRP assessment

Breeding population
A species specific PVA is not available. The upscaled MVP is estimated at 12,500 pairs. The Baltic Sea population of *C. a. schinzii* encompassed 462-573 breeding pairs in 2008- 2012 (Table 4.4.2; European Environment Agency 2015). This is far below the upscaled MVP (12,500 pairs). The current breeding population is considered depleted. Exact historical numbers are not available, but the subspecies should have been far more numerous, as densities up to 50 pairs per km^2 were recorded there (Hagemeijer & Blair 1997). The depletion took place before 1980. At that time the population was likely to be around 1000 pairs. This is far below the upscaled MVP (12,500 pairs) as well. So the FRP should be much more than 12,500 pairs.

Passage/wintering population
During passage and wintering some of the flyway populations mix and cannot be told apart in counts. Therefor we subdivided de passage wintering populations in two: a West- and an East-European.

Example: The West European (including North Africa) passage/wintering population consists of breeding birds from *C. a. alpina* from northern Europe and *C. a. schinzii* from both the British Isles and Baltic breeding population. Based on winter counts the numbers in West-Europe have decreased slightly since 1980. The numbers are dominated by *C a. alpina* (Table 2): around 1,330,000 birds (including North Africa) recently. Time series going back to 1990 show peak numbers of 1,500,000 birds (van Roomen *et al.* 2015). So the FRP should be at least 1,500,000 birds for the West-European wintering population including North-Africa.

EU national targets should be based on the carrying capacity of sites/habitat. Numbers can be used as a proxy but care should be taken because they have to reflect the carrying capacity. Note that numbers are subject to fluctuations because of climate conditions.

Conclusions

Baltic area (breeding population *C. a. schinzii*): FRP >> 12,500 pairs.

Western Europe + North Africa (wintering population): FRP ≥ 1,500,000 birds.

Step 2.2 - FRR assessment

Breeding population

The Baltic population of *C. a. schinzii* shows a slight decline in distribution between the first European Breeding Bird Atlas and the Bird Directive reporting 2008-2012, however not as strong as the decline in numbers. We propose to set FRR for the Baltic breeding population of *schinzii* at least at the level of the first European Breeding Bird Atlas: c 100 occupied 50x50 km squares = 25,0000 km^2.

Passage/wintering population

Accumulated distribution over several years, derived from maps including the distribution of wintering populations in peak years can be used as ‘current value’ for the Dunlin’s supranational FRR’s.

Conclusions

Baltic area (breeding population C. a. schinzii): FRR ≥ 25,0000 km2 (at 50x50 km squares level).

Westen Europe + North Africa (wintering population): The accumulated distribution over several years, derived from maps including the distribution of wintering populations in peak years can be used as ‘current value’ for the Dunlin’s supranational FRR.

References

BirdLife International (2017) Species factsheet: Calidris alpina. Downloaded from http://www.birdlife.org on 08/03/2017.

European Environment Agency 2015: https://bd.eionet.europa.eu/article12/summary?period=1&subject=A332

Delany, S., Scott, D., Dodman, T. & Stroud, D. (eds) 2009. An Atlas of Wader Populations in Africa and Western Eurasia. Wetlands International. Wageningen, The Netherlands.

Hagemeijer E.J.M. & Blair M.J. (eds.) 1997. The EBCC Atlas of the European Breeding Birds: Their Distribution and Abundance. T. & A. D. Poyser, London.

Van Roomen M., Nagy S., Foppen R., Dodman T., Citegetse G. & Ndiaye A. 2015. Status of coastal waterbird populations in the East Atlantic Flyway. With special attention to flyway populations making use of the Wadden Sea. Programme Rich Wadden Sea, Leeuwarden, The Netherlands, Sovon, Nijmegen, The Netherlands, Wetlands International, Wageningen, The Netherlands, BirdLife International, Cambridge, United Kingdom & Common Wadden Sea Secretariat, Wilhelmshaven, Germany.

4.5 White-tailed Eagle (*Haliaeetus albicilla*) in Europe and the EU

André van Kleunen, Marc van Roomen & Ruud Foppen

Step 1.1 - Biology of the species
The White-tailed Eagle is a large raptor (length 77-92 cm, wingspan >2 m, body mass 0.4-0.6 kg), with broad board shaped wings, a protruding head and relatively short wedge-shaped tail and a predominantly dark brownish plumage.

The species requires large and open expanses of lake, coast or river valley, within the boreal, temperate and tundra zones, nearby to undisturbed cliffs or open stands of large, old-growth trees for nesting. Its food is vertebrates (fish, mammals and especially birds), from marine, freshwater and terrestrial environments. Most of the European White-tailed Eagles are considered sedentary, apart from a part of the population in Fennoscandia and large parts of Russia. Birds are usually seen singly, or in twos or threes. The nest is a huge construction of sticks and branches usually built in trees, at cliffs and even in pylons. Clutch size: usually two eggs. Breeding failure can be high: average 0,2-1,1 fledged young per breeding pair. They reach sexual maturity after five years. They can live to 27 years in the wild. The nest is built within 10 km of the foraging grounds. Breeding densities are usually low but in good habitat occupied nests can be found at 400 m distance of each other (sources: BirdLife International 2017; Del Hoyo *et al.* 1994, Clutz von Blotzheim *et al.* 1989).

Step 1.2 - Spatial scale of functioning

Breeding population
The breeding range of the White-tailed Eagle extends from Northern-Europe, across Central Europe to South-east Europe (Figure 1). The species is absent from most parts of western and southern Europe. The EU27 population is estimated to number 3,500-4,300 breeding pairs, (European Environment Agency 2015). In particular in central and southeast Europa its distribution becomes fragmented. Relatively isolated populations occur in Scotland (reintroduced) and Iceland (Figure 4.5.1).

The species' dispersal capacity is estimated at 500 km. This means that at least all continental breeding birds from Northern Scandinavia up to Greece should be considered as one metapopulation despite the apparently fragmented distribution of the species. Given the overall sedentary nature, this corresponds to population category S1: widespread sedentary species with more or less continuous distribution (often crossing national boundaries) and populations (assessment units) with more or less exchange at or below national level. Although in some cases home ranges can be quite large – foraging up to 10 km away from the nest has been recorded – generally nesting and foraging grounds are not that separated (not trans-national) and closer to each other, so that category S4 (animal species with individuals with large home ranges >100 up to >1000 km^2) is not relevant.

Passage/wintering population
Wintering areas are located in Northwest, Central and Southeast Europe. These are highly fragmented, but rather often located in breeding areas of the species. Since only part of the population is migratory and a significant part of its winter range overlaps with the breeding range, we propose not to consider the wintering population of White-tailed Eagle separately with regard to setting FRVs.

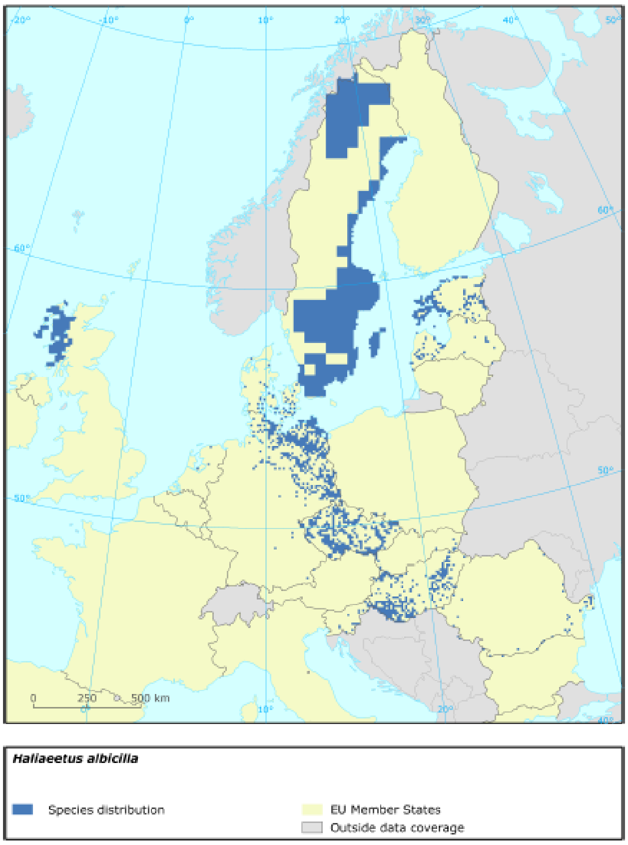


Figure 4.5.1 *Breeding distribution of White-tailed Eagle in EU27 in 2008 -2012 (10 km2-grid cells). Source: European Environment Agency (2015)*

Conclusions

The breeding range of the White-tailed eagle extends from Northern-Europe across Central Europe to Southeastern Europe. Given its large estimated dispersal capacity this range seems one metapopulation. Although the species is migratory in Northern-Europa, wintering grounds largely correspond with breeding grounds. Therefore we propose to set FRVs for the breeding population only following category S1 (at the national level).

Step 1.3 - Historical perspective: what happened to the species?

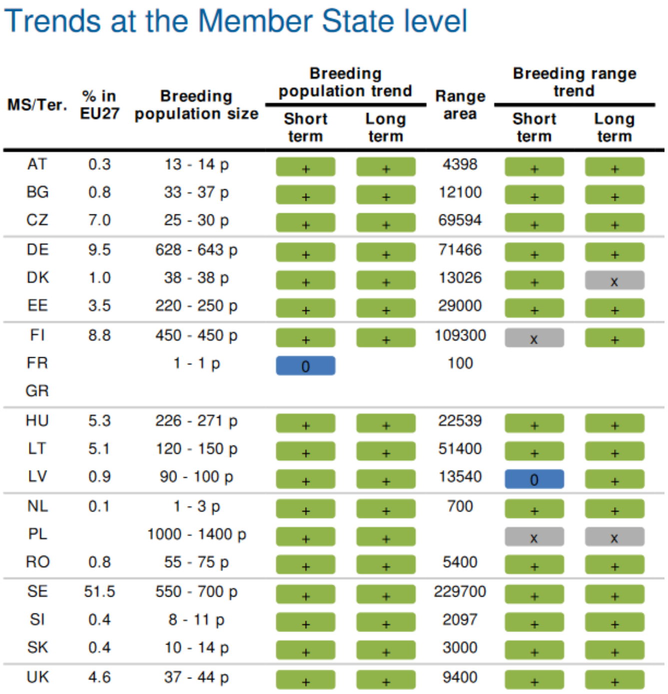
Until the early 20th century the species was heavily persecuted and exterminated from parts of Europe. After protection measures had been taken the population started growing from refuges in Scandinavia and Eastern Europe. This population growth was reversed temporally in the period 1950-1970 as a consequence of poisoning by chemical pollutants, in particular DDT and PCBs (Hagemeijer & Blair 1997). During the last decades the population has recovered as a result of measures such as protecting eyries, providing safe (non-poisoned) food and by re-introductions to certain areas such as Bavaria (BirdLife International 2017).

Step 1.4 - Analysis of distribution and trends

Distribution data and data on numbers are available from Article 12 Bird Directive reporting 2008-2012 (European Environment Agency 2015) and Hagemeijer & Blair (1997). The population has increased in numbers and distribution in Europe since 1980. At country level at least all EU27 countries harbouring White-tailed Eagles show positive trends since 1980 (Table 4.5.1). The EU-breeding population is estimated at 3500-4300 pairs in 2008-2012 (European Environment Agency 2015). Europe wide the population is estimated at 9000-12,300 pairs (BirdLife International 2017) compared to only 2400 pairs in 1980 (Hagemeijer & Blair 1997).

Numbers in some countries are still quite low but further recovery of the population seems likely e.g. the population is expanding in the West in the Netherlands.

Table 4.5.1 ***Population statistics of White-tailed Eagle in EU-member states for the periode 2008-2012 (European Environment Agency 2015). Short term trend considers the last 10 years and long term trend since the period since 1980***



Trends at the Member State level

MS/Ter.	% in EU27	Breeding population size	Breeding population trend Short term	Breeding population trend Long term	Range area	Breeding range trend Short term	Breeding range trend Long term
AT	0.3	13 - 14 p	+	+	4398	+	+
BG	0.8	33 - 37 p	+	+	12100	+	+
CZ	7.0	25 - 30 p	+	+	69594	+	+
DE	9.5	628 - 643 p	+	+	71466	+	+
DK	1.0	38 - 38 p	+	+	13026	+	x
EE	3.5	220 - 250 p	+	+	29000	+	+
FI	8.8	450 - 450 p	+	+	109300	x	+
FR		1 - 1 p	0		100		
GR							
HU	5.3	226 - 271 p	+	+	22539	+	+
LT	5.1	120 - 150 p	+	+	51400	+	+
LV	0.9	90 - 100 p	+	+	13540	0	+
NL	0.1	1 - 3 p	+	+	700	+	+
PL		1000 - 1400 p	+	+		x	x
RO	0.8	55 - 75 p	+	+	5400	+	+
SE	51.5	550 - 700 p	+	+	229700	+	+
SI	0.4	8 - 11 p	+	+	2097	+	+
SK	0.4	10 - 14 p	+	+	3000	+	+
UK	4.6	37 - 44 p	+	+	9400	+	+

Conclusions

The population of White-tailed Eagle in the EU has increased and expanded in the last decades. In some countries the numbers still seem quite low, further increase and spread seems likely.

Step 2.1 - FRP assessment

A species specific PVA is not available. The MVP is estimated at 2500 pairs (see § 5.4). This number is clearly exceeded for the European population: 9000-12,300 pairs and even for the EU-countries only (3500-4300 pairs) but not so in the eighties when the European population was estimated at only 2400 pairs). Since the species is recovering from a long term decline across its range, we propose to use FRP > current value awaiting a population equilibrium at EU and national level. In some countries numbers are still quite low and there seems further recovery ahead.

Conclusions

FRP (generally at MS level) > CV.

Step 2.2 - FRR assessment

The EU-wide recovery of the species also applies to its range and distribution (European Environment Agency 2015). As for the FRP, we propose to use FRR > current value awaiting a population equilibrium at EU and national level. In some countries distribution is still quite small and there seems further recovery (expansion) ahead.

Conclusions

FRR (generally at MS level) > CV.

References

BirdLife International (2017) Species factsheet: Haliaeetus albicilla. Downloaded from http://www.birdlife.org on 08/03/2017.

Clutz von Blotzheim U.N., Bauer K.M. & Bezzel E. 1989. Handbuch der Vögel Mitteleuropas. Band 4 Falconiformes. Aula Verlag, Wiesbaden.

European Environment Agency 2015: https://bd.eionet.europa.eu/article12/summary?period=1&subject=A332

Hagemeijer E.J.M. & Blair M.J. (eds.) 1997. The EBCC Atlas of the European Breeding Birds: Their Distribution and Abundance. T. & A. D. Poyser, London.

Del Hoyo J., Elliott A., Sargatal J. (eds.) 1994. Handbook of the Birds of the World. Vol 2. New world Vultures to Guinea Fowl, Lynx Edicions, Barcelona.

4.6 Wood Warbler (*Phylloscopus sibilatrix*) in Sweden

Wouter Langhout, Ariel Brunner & Ivan Ramírez

Step 1.1 - Biology of the species

Wood Warbler is a small, fully migratory passerine (4 – 15 g; 12 cm) (Carboneras *et al.* 2016). The species is widespread; in 2007-2012 the species was breeding in all EU Member States except Portugal, Malta and Cyprus (BirdLife International 2015). The species is as result present in most biogeographical regions, but is less common in the Mediterranean biogeographical region and absent in Macaronesia. Globally, the distribution of the breeding populations of the species is restricted to Europe and part of the Middle East (BirdLife International 2017). The species winters in sub-Saharan Africa (Snow and Perrins 1998). In its African non-breeding grounds it uses humid evergreen forest, moist thickets, forest edge, dry woodland and forested mountain slopes as well as wooded savanna and scattered trees in forest clearings, often in fig trees (*Ficus*), occasionally in mangroves (Carboneras *et al.* 2017). The Wood Warbler breeds in lowlands, in moist and shady deciduous woods, typically beech (Fagus), mixed oak (*Quercus*), hornbeam (*Carpinus*) and sweet chestnut (*Castanea*), with closed canopy and sparse undergrowth. It is also found in mixed stands of spruce (*Picea*), alder (*Alnus*), birch (*Betula*), pine (*Pinus*) or occasionally ash (*Fraxinus*).The optimal breeding areas are mixed stands with trees of mixed ages spaced well apart. It breeds from May to July and normally lays five to seven eggs. The female chooses the nest site and builds the nest which is a ball of dry grasses, leaves, plant stems and fibres, bark strips and animal hair. It is usually built on the ground but occasionally found under a fallen tree or roots (Carboneras *et al.* 2016). Adults also rarely return to breeding sites of the previous year, instead showing nomadic behaviour and selecting breeding sites on the basis of rodent densities (Wesołowski *et al.* 2009, Pasinelli *et al.* 2016). Given the densities of territories observed in good quality habitats, home ranges are around 3 hectare (Wesołowski et al 2002, Bibby *et al.* 1989). The diet is mainly insects and other invertebrates, with some fruit taken in the autumn. The mean natal dispersal distance is estimated to be 20 km (Jiguet *et al.* 2007).

In some breeding areas changing forestry management practices have caused declines in Wood Warbler populations (Mallord *et al.* 2012). Wood Warblers, like many other long-distance migratory insectivorous passerines, have been declining since 1970 (Sanderson *et al.* 2006). While the threats to specific species on the wintering grounds are poorly understood, potential candidates are habitat loss and degradation, changing rainfall patterns, climate change and potentially that long-distance migrants are more vulnerable to changes in their breeding grounds compared to sedentary species (Sanderson *et al.* 2006).

Step 1.2 - Spatial scale of functioning

Breeding population

It is likely that most Wood Warblers belong to one large continuous European population, given the continuous and widespread distribution. Most of the distribution lies within five mean natal dispersal distances (100 km) from each other. There is possibly an isolated breeding population in Greece (Crete). The breeding population is very large, there are 3,890,000-6,180,000 breeding pairs in the EU (BirdLife, 2015). The European breeding population exceeds by far the size for which population viability aspects would need to be taken into account (12 500 breeding pairs).

Passage/wintering population

Due to the large population and continuous distribution, and the fact that the species winters outside Europe there is no added value in defining FRP and FRR for wintering populations.

Conclusions

Population category MR1 should be applied, as the Wood Warbler is a migratory species with a widespread and continuous distribution, with exchange below Member State level. As a result, FRVs should be assessed at the national level.

Step 1.3 - Historical perspective: what happened to the species?

The long-term trend (since about 1980) of the EU population is negative, while the short-term trend (since about 2000) is stable. The population status in the EU is 'Depleted' (EEA 2015), due to its long-term decline and recent stabilisation. There is no detailed information on the historic population trend, although the forest area in the EU has considerably changed since 1900 due to re- and afforestation as well as regional deforestation (Fuchs *et al.* 2015), probably resulting in an overall increase in habitat. The species could also have been affected by historic land use changes in sub Saharan Africa.

Step 1.4 - Analysis of distribution and trends

Distribution data and data on numbers are available from Article 12 Bird Directive reporting 2008-2012 (EEA 2015) and Hagemeijer & Blair (1997). Breeding population estimates of medium or good quality are available for all EU Member States, except for Croatia, Greece, Italy and Slovakia where only poor estimates are available (BirdLife International 2015).
Negative trends in population size have been reported for Belgium (long-term), Germany (short-term and long-term), Denmark (long-term), Estonia (long-term), Finland (short-term and long-term), France (long-term), Ireland (short-term and long-term), Luxembourg (long-term), Netherlands (long-term), Sweden (long-term), and United Kingdom (short-term and long-term) (BirdLife International 2015).

Negative trends in breeding range have been reported in Belgium (short-term and long-term), Denmark (short-term), France (long-term), Ireland (short-term) Netherlands (short-term and long-term) and United Kingdom (short-term and long-term) (BirdLife International 2015).

The species also has a (climate) species distribution model, with a very good fit (Huntley et al 2007). The species breeding distribution is predicted well by the annual temperature sum, the coldest month mean temperature and the seasonal moisture deficiency.

Conclusions

The long-term trend of the EU population is negative, while the short-term trend is stable. The population status in the EU is 'Depleted'. Wood Warbler is declining in population and or breeding range in several EU Member States on the long-term and in some countries also on the short-term.

There is a very good species distribution model available and there are good or medium quality estimates for breeding populations in most EU Member States.

Step 2.1 - FRP assessment (Sweden)

For widespread and common species, the methodology of Brambilla *et al.* (2011) and Brambilla *et al.* (2014) can be applied, combined with estimates of area of suitable good and average habitat. The Wood Warbler is predominantly a species of deciduous broadleaf forest (MAES category G1), which is considered good habitat for the purpose of identifying a FRP. Good densities in broadleaf forests are in the order of 3,0 breeding pairs per 10 hectares (Wesołowski et al 2002, Bibby *et al.* 1989). According to the MAES Map of Ecosystem types (EEA 2015) there is 25 000 km^2 of deciduous broadleaf forest in Sweden. However, analysis using ArcGIS showed that only 13 000 km^2 meets the climatic requirements of the species, with the remainder being unsuitable as evident from the predicted distribution under the present day climate (Huntley *et al.* 2007) and the current distribution (Hagemeijer & Blair 1997). Given these estimates, potentially there is room for 390 000 breeding pairs of Wood warblers in Sweden in broadleaf forest. The species also occurs in mixed forest (MAES category G4) which is considered average habitat for the purpose of identifying a FRP. Good densities in average habitat are around 0,3-0,6 breeding pairs per 10 hectares, based on the median of densities across habitats (Bibby 1989, Herremans 1993). There is 15 000 km^2 that meets the climatic requirements of the species. Potentially there is room for up to 90 000 breeding pairs in mixed forests. The predicted distribution of the species can be improved by including further variables on the basis of the ecology of the species. Potential candidates for the Wood Warbler case are soil type and altitude. This is without prejudice to future changes in broadleaf and mixed forest area and forest type (e.g. due to succession or restoration) and climate change. Identifying an FRP is ideally done on the basis of a spatially explicit scenario for the habitat of the species, which includes not only the current and future habitat under climate change, but also other relevant factors such as changes in land use and the results of implementation of other EU environmental legislation.

The current methodology assumes that all broadleaf forest and all mixed forest is suitable for Wood Warbler. This assumption may not be met, as there are inherent uncertainties in climatic requirements, suitability of habitat and accuracy of the habitat maps at EU level, as well as well as a limited resolution of the MAES ecosystem types. It is therefore suggested to not use the full value, but instead as a rule of thumb apply 90% of the value to pre-empt the inherent uncertainties. This rule of thumb resembles the 90% threshold of habitat area in good condition for assessing 'structure and functions' of habitat types[1].

The FRP for Wood Warbler in Sweden is therefore preliminary estimated at 430 000 breeding pairs. In 2007-2012 there were an estimated 141 000-292 000 breeding pairs in Sweden (BirdLife International 2015) down from an estimated 176 000 - 424 000 breeding pairs in 1980 (back-casted numbers on the basis of BirdLife International 2015). The species would have an 'unfavourable conservation status' regarding its population in Sweden on the basis of this estimate. On the basis of the estimated long-term trend, the species may have been close to favourable in 1980.

Conclusions

The FRP for the Wood Warbler in Sweden, a common and widespread species, can be estimated on the basis of good densities in broadleaf forest and average densities in mixed forests:
FRP = 430 000 breeding pairs.

Step 2.2 - FRR assessment (Sweden)

The climatic model (Huntley *et al.* 2007) shows that on the basis of the relevant climatic factors most of Sweden meets the climatic requirement of the Wood Warbler, with the exception of the far north. Throughout the part of Sweden with the climatic envelope, the habitat of the species (broadleaf or mixed forest) is present and therefore the Wood Warbler should have a continuous range throughout Sweden (without the need to take into account habitat that may have been lost in recent or historic times). For birds, a gap of 50 km is considered to fall within a continuous range (N2K Group, 2011). For the current analysis, therefore all gaps within the climate envelope were closed using ArcGIS.

This results in a Favourable Reference Range of around 370 000 km^2 (82% of the European Breeding Bird Atlas 50 x 50 km squares). The current range is around 370 000 km^2 (84% of the European Breeding Bird Atlas 50 x 50 km squares) and therefore the species would have a 'favourable conservation status' regarding its range in Sweden under this methodology.

Conclusions

Most of Sweden (except the far north) has a suitable climate and suitable habitat for the Wood Warbler and therefore should be part of the Favourable Reference Range:
FRR = 370 000 km^2 (82% European Breeding Bird Atlas 50 x 50 km squares).

References

Bibby, C. J. (1989). A survey of breeding Wood Warblers Phylloscopus sibilatrix in Britain, 1984–1985. Bird Study, 36(1), 56-72.

BirdLife International (2015). European Red List of Birds. Luxembourg: Office for Official Publications of the European Communities. Including species factsheets.

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[1] DG Environment. 2017. Reporting under Article 17 of the Habitats Directive: Explanatory notes and guidelines for the period 2013-2018. Brussels.

BirdLife International (2017) IUCN Red List for birds. Downloaded from http://www.birdlife.org on 25/08/2017.

Brambilla, M., Gustin, M., & Celada, C. (2011). Defining favourable reference values for bird populations in Italy: setting long-term conservation targets for priority species. Bird Conservation International, 21(1), 107-118.

Brambilla, M., Celada, C., & Gustin, M. (2014). Setting Favourable Habitat Reference Values for breeding birds: general principles and examples for passerine birds. Bird Conservation International, 24(3), 263-271.

Carboneras, C., *et al.* (2016). Handbook of the Birds of the World Alive. Lynx Edicions

EEA (2015). State of Nature in the EU. Luxembourg: Office for Official Publications of the European Union, 2015.

Fuchs, R., Herold, M., Verburg, P. H., Clevers, J. G., & Eberle, J. (2015). Gross changes in reconstructions of historic land cover/use for Europe between 1900 and 2010. Global change biology, 21(1), 299-313.

Jiguet, F., Gadot, A. S., Julliard, R., Newson, S. E., & Couvet, D. (2007). Climate envelope, life history traits and the resilience of birds facing global change. Global Change Biology, 13(8), 1672-1684.

Hagemeijer, W. J., & Blair, M. J. (1997). The EBCC atlas of European breeding birds. T & AD Poyser, London.

Herremans, M. (1993). Clustering of territories in the Wood Warbler Phylloscopus sibilatrix. Bird Study, 40(1), 12-23.

Huntley, B., Green, R. E., Collingham, Y. C., & Willis, S. G. (2007). A climatic atlas of European breeding birds. Barcelona: Lynx Edicions.

Mallord, J.W., Charman, E.C., Cristinacce, A. and Orsman, C.J. 2012. Habitat associations of Wood Warblers Phylloscopus sibilatrix breeding in Welsh oakwoods. Bird Study, 59(4): 403-415.

N2K Group (2011) Assessment and reporting under Article 12 of the Birds Directive Explanatory Notes & Guidelines for the period 2008-2012. Final Version, Compiled by the N2K Group under contract to the European Commission.

Pasinelli, G., Grendelmeier, A., Gerber, M., & Arlettaz, R. (2016). Rodent-avoidance, topography and forest structure shape territory selection of a forest bird. BMC ecology, 16(1), 24.

Sanderson, F. J., Donald, P. F., Pain, D. J., Burfield, I. J., & Van Bommel, F. P. (2006). Long-term population declines in Afro-Palearctic migrant birds. Biological conservation, 131(1), 93-105

Snow, D.W. and Perrins, C.M. 1998. The Birds of the Western Palearctic vol. 2: Passerines. Oxford University Press, Oxford

Traill, L.W., C.J.A. Bradshaw & B.W. Brook. 2007. Minimum viable population size: A meta-analysis of 30 years of published estimates. Biological Conservation 139: 159–166.

Wesołowski, T., Tomiałojć, L., Mitrus, C., Rowiński, P., & Czeszczewik, D. (2002). The breeding bird community of a primaeval temperate forest (Białowieża National Park, Poland) at the end of the 20th century. Acta ornithologica, 37(1), 27-45

Wesołowski, T., Rowiński, P., & Maziarz, M. (2009). Wood Warbler Phylloscopus sibilatrix: a nomadic insectivore in search of safe breeding grounds?. Bird Study, 56(1), 26-33.

4.7 Northern Gannet (*Morus bassanus*) in Europe

Wouter Langhout, Ariel Brunner & Ivan Ramírez

Step 1.1 - Biology of the species
Nothern Gannet is a large seabird (87–100 cm; 2,300–3,600 g) (Carboneras *et al.* 2016). The species is fully migratory (BirdLife International 2017).

The species has a limited breeding distribution in the EU. In 2007-2012 it was breeding in 4 Member States: France, Germany, Ireland, and the United Kingdom (BirdLife International 2015). The species is as a breeding bird restricted to the Atlantic Biogeographic Region, although the species irregularly breeds in the Mediterranean in France (Deideri *et al.* 2014) and Italy (Giagnonin *et al.* 2015) Globally the species is restricted to Europe and North America (Canada) (BirdLife International 2017). There are three distinct wintering areas for European birds 1) the North Sea and the English Channel, 2 the Irish sea, the Bay of Biscay, and the English Channel and 3) and the coast of North Africa) (Fort *et al.* 2012). Area 1) and 2) overlap in the Channel. Age is also a determinant for migration, young birds will migrate to the extreme south of its range, whereas adults range less extensively, but still regularly winter in the Mediterranean (del Hoyo *et al.* 1992). Different populations also have different wintering grounds, with northern populations also wintering further north (Fort *et al.* 2012) Breeding is highly seasonal starting between March and April, usually in large colonies on cliffs and offshore islands, but also sometimes on the mainland. Home ranges (territories) have not been quantified for these species, but are very large (in the order of several thousand km2, given the large foraging distance (next paragraph)

This strictly marine species forages mostly over continental shelves, feeding on shoaling pelagic fish which are mostly caught by plunge-diving from large heights. It also attends trawlers and will form large congregations where food is plentiful. Foraging distances are up to 540 km, potentially more (Hamer *et al.* 2000). The natal dispersal distance is expected to be around 1,400 – 1,800 km given the species weight and diet. There is likely a substantial exchange between all EU colonies (Votier *et al.* 2011, Moss *et al.* 2002).

Overfishing and prey depletion is likely to affect this species, although populations in the U.K. and Ireland were not seriously affected by Shetland sandeel stock crash in the mid-1980s (Carboneras *et al.* 2014). Incidental capture in fishing gear, including in longlines and purse seines also poses a threat; and the species was the most commonly caught along the Portuguese mainland coast (Oliveira *et al.* 2015). The species is hunted for food in some places, for example, a small annual harvest is carried out on Sula Sgeir, off northwest Scotland. The small population of northern Norway has suffered local declines and extinctions thought to be mainly due to harassment by White-tailed Eagles (*Haliaeetus albicilla*). The species is vulnerable to collisions with offshore wind turbines (Bradbury *et al.* 2014).

Step 1.2 - Spatial scale of functioning

Breeding population
It is likely that all Northern Gannets belong to one large continuous European Population, given the natal dispersal distance and the likely exchange between colonies (see above). The breeding population is very large, there are 281,000 breeding pairs in the EU (BirdLife International 2015). The European breeding population exceeds by far the size for which population viability aspects would need to be taken into account (2500 breeding pairs).

Passage/wintering population
As mentioned above the wintering location of Northern Gannet is dependent on age and population of origin, and winter in three distinct areas: 1) the North Sea, 2) the Irish Sea, English Channel, and Bay of Biscay and 3) the coast of North West Africa. It is therefore recommended to define a FRP and a FRR for at least the North Sea and Irish Sea wintering population as this falls within the jurisdiction of the EU.

Conclusions

Population category MR4 should be applied for the European breeding population, as the Northern Gannet is a migratory species with a limited distribution and a large home range, with exchange above Member State level. As are result, FRP and FRR should be assessed at supranational level.
For the passage/wintering population we recommend to set a supranational FRP and FRR for the distinct North Sea and Irish Sea populations (category MNR4).

Step 1.3 - Historical perspective: what happened to the species?
The long-term trend (since about 1980) and the short-term trend (since about 2000) is increasing (BirdLife International 2015). The population status in the EU is 'Secure' (EEA 2015). In historical perspective, the trend is likely still negative, as the species has been heavily hunted from at least the 16th to the 20th century, with some colonies maximally exploited and entire colonies have been lost (Gannet Stone, Calf of Man, Isle of May Nelson 2002). The global population in 1900 was no more than 50 000 breeding pairs (Nelson 2002) and according to one source (cited in Nelson 2002) the global population was reduced by two thirds over the 19th century. Legal protection in the UK was put in place in 1869 (the Seabirds Preservation Act), protecting adults from hunting everywhere except on St Kilda (Barclay-Smith, 1959).

Step 1.4 - Analysis of distribution and trends
Distribution data and data on numbers are available from Article 12 Bird Directive reporting 2008-2012 (EEA 2015) and Hagemeijer & Blair (1997).

Breeding population
Breeding population estimates of good quality are available for all EU Member States where the species is breeding (BirdLife International 2015). The species also has a (climate) species distribution model, with a 'poor fit' (Huntley et al 2007). The species has a very scattered coastal distribution, with major deficiencies in the predicted distribution. There are no recent negative trends in population size on Member State level. The United Kingdom reported a negative short-term trend in distribution (EEA 2015). However, historical population numbers before 1800 were possibly higher than today.

Passage/wintering population
The non-reproductive populations in the North Sea and Irish Sea have not been monitored as such, and as a result only proxies would be available on the basis of the breeding populations.

Step 2.1 - FRP assessment

Breeding population
A full population model is available for Gannet in the United Kingdom and Ireland (Wildfowl and Wetlands Trust 2012). The model indicates a carrying capacity of 950 000 – 1 250 000 individuals, under the assumption that density dependence exists. At carrying capacity the population has a stable age-distribution (the end result of the model runs on the basis of the parameters for reproduction and survival from one age-cohort to the next). In the absence of detailed field data, carrying capacity (i.e. the maximum attainable population) can be used as a proxy of the species being in good density in its existing habitat. This is under the assumption that the current pressures and threats are not lowering the carrying capacity significantly.

The population model could be further improved by incorporating information on the future development of fish stock, in particular attaining Maximum Sustainable Yield under the Common Fisheries Policy, and the incidental bycatch in fisheries, as these would address the widespread pressures and threats and would make the use of carrying capacity as a proxy for FRP more robust. If the stable age-distribution is approximately the same as the current stable age-distribution (Wildfowl and Wetlands Trust 2012), 60% of the individuals would be in the reproductive cohort, which can be assumed to consist of 100% breeding pairs. This results in 225 000 – 300 000 breeding pairs in the UK and Ireland. This would be most of the European population, except France, Iceland, Norway and small populations in Russia and Germany. The current methodology relies heavily on the population model, and on attaining carrying capacity which may not be a stable equilibrium. It is therefore suggested to not use the full carrying capacity, but instead as a rule of thumb apply 90% of the value to generate a robust and achievable value. This rule of thumb resembles the 90% rule of thumb for assessing 'structure and functions' of habitat types[2].

The FRP for the European breeding population of Gannet is therefore estimated to be in the order of 300,000 breeding pairs. In order to generate a more accurate FRP of the population model would be needed to determine exact carrying capacity, to generate a stable age-distribution, and to include the populations in France, Germany, Iceland, Norway, and Russia. The European breeding population of Northern Gannet is 263,000 breeding pairs (BirdLife International, 2015). This population would very likely be assessed as having a 'favourable conservation status' regarding its European breeding population or approaching that level.

Passage/wintering population

Due to the absence of monitoring data of the wintering population, the FRP for the 1) North Sea and English Channel and 2) Irish Sea, English Channel, and Bay of Biscay wintering populations can only be identified on the basis of the breeding populations that winter in the areas. On the basis of a stable-age distribution of the adult cohort, FRPs can be estimated for the North Sea on the basis of the Norwegian colonies and the Irish Sea on the basis of the UK breeding colonies.

Conclusions

A supranational FRP for the European breeding population of the Northern Gannet can be identified through the use of carrying capacity as a proxy for FRP, on the basis of a population model:
FRP is in the order of 300 000 breeding pairs.

The population model needs to be updated to include the populations outside the United Kingdom and Ireland, and needs to include the effect of fisheries and incidental bycatch in order to obtain a robust FRP estimate.

The FRPs for the for the North Sea and Irish Sea wintering populations could be estimated using the population model for the European breeding population, based on respectively the Norwegian breeding colonies and the UK breeding colonies.

Step 2.2 - FRR assessment

Breeding population

Due to the patchy breeding distribution, the climatic model (Huntley *et al.* 2007) has only a poor fit, and should not be used for identifying FRR. The species has expanded its breeding range in the long-term, due to the establishment of new colonies (BirdLife International 2015). At the moment, adult survival is limiting population growth rather than availability of suitable nesting habitat. Although it cannot be ruled out that new colonies may still be established, the FRR occupied under FRP will likely to be very close to the current value. The loss of colonies on the islets Gannet Stone, Calf of Man and Isle of May in historic times (before 1900) may be still reversible, if the habitat is restored and invasive alien species are removed, although the recolonization of Northern Gannets may even not happen then.

Conclusions

The FRR is likely currently sufficient to accommodate the FRP and the species has returned to most of its historic range, with the exception of some small islets:
FRR = CV.

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[2] DG Environment. 2017. Reporting under Article 17 of the Habitats Directive: Explanatory notes and guidelines for the period 2013-2018. Brussels.

References

Barclay-Smith, P. (1959). The British contribution to bird protection. Ibis, 101(1), 115-122.

BirdLife International (2015). European Red List of Birds. Luxembourg: Office for Official Publications of the European Communities. Including species factsheets.

BirdLife International (2017) IUCN Red List for birds. Downloaded from http://www.birdlife.org on 25/08/2017.

Bradbury, G., Trinder, M., Furness, B., Banks, A.N., Caldow, R.W.G. and Hume, D. 2014. Mapping Seabird Sensitivity to Offshore Wind Farms. PLoS ONE 9(9): e106366.

Deideri J., Deideri M., Beyney M.-C., Rouger M.-H. (2014). Dix ans de suivi des Fous de Bassan de la Côte Bleue. Faune PACA Publ. 43: 1-21

EEA (2015). State of Nature in the EU. Luxembourg: Office for Official Publications of the European Union, 2015.

Fort, J., Pettex, E., Tremblay, Y., Lorentsen, S. H., Garthe, S., Votier, S., ... & Bearhop, S. (2012). Meta-population evidence of oriented chain migration in northern gannets (Morus bassanus). Frontiers in Ecology and the Environment, 10(5), 237-242.

Giagnoni, R., Conti, C. A., Canepa, P., & Nardelli, R. (2015). First breeding records of Northern Gannet Morus bassanus in Italy. Avocetta, 39, 93-95.

Hamer, K. C., Phillips, R. A., Wanless, S., Harris, M. P., & Wood, A. G. (2000). Foraging ranges, diets and feeding locations of gannets Morus bassanus in the North Sea: evidence from satellite telemetry. Marine ecology progress series, 257-264.

Huntley, B., Green, R. E., Collingham, Y. C., & Willis, S. G. (2007). A climatic atlas of European breeding birds. Barcelona: Lynx Edicions.

Moss, R., Wanless, S., & Harris, M. P. (2002). How small northern gannet colonies grow faster than big ones. Waterbirds, 25(4), 442-448.

Oliveira, N. Henriques, A., Miodonski, J., Pereira, J., Marujo, D., Almeida, A., Barros, N., Andrade, J., Marçalo, J., Santos, I., Oliveira, B., Ferreira, M., Araújo, H., Monteiro, S., Vingada, J. and Ramírez, I. 2015. Seabird Bycatch in Portuguese Mainland Coastal Fisheries: An Assessment through on-Board Observations and Fishermen Interviews. Global Ecology and Conservation 3: 51-61.

Votier, S. C., Grecian, W. J., Patrick, S., & Newton, J. (2011). Inter-colony movements, at-sea behaviour and foraging in an immature seabird: results from GPS-PPT tracking, radio-tracking and stable isotope analysis. Marine biology, 158(2), 355-362.

Wildfowl and Wetlands Trust (2012) Demographic data, population model and outputs - Gannet Population Viability Analysis. Project SOSS-04. Slimbridge, United Kingdom

4.8 Great White Egret (*Ardea alba*) in Europe

Wouter Langhout, Ariel Brunner & Ivan Ramírez

Step 1.1 - Biology of the species
All-white heron; 80 – 104 cm; 700 – 1,700 g; wingspan 140–170 cm. (Carboneras *et al.* 2016). Palearctic populations are migratory (Flint *et al.* 1984, Martínez-Vilalta *et al.* 2014). All populations of Great White Egret undergo post-breeding dispersive movements (Martínez-Vilalta *et al.* 2014).

In 2007-2012 the species was breeding in the EU in 14 Member States, mainly in West, Central and South-East Europe in the Continental, Pannonian, Black Sea, Mediterranean and Atlantic biogeographic regions (EEA 2015). The species was also present as a wintering bird in 12 Member States, broadly speaking in the same biogeographic regions Great White Egrets have in the 1980s shifted their winter areas away from North Africa and the Eastern Mediterranean, and now more and more winter near their breeding sites (Lawicki 2014). Globally, the species has a cosmopolitan distribution (BirdLife International 2017). The timing of the breeding season varies geographically (Martínez-Vilalta *et al.* 2014) although temperate breeders tend to nest in the spring and summer (Kushlan and Hancock 2005). The species typically breeds in colonies of tens, hundreds or even a thousand pairs (Kushlan and Hancock 2005, Martínez-Vilalta *et al.* 2014), sometimes with other species. Some populations also show a tendency to breed solitarily or in small groups (Martínez-Vilalta *et al.* 2014). The nest is constructed from sticks (Kushlan and Hancock 2005) and vegetation (Brown *et al.* 1982) and is normally positioned over water at a height of 1–15 m (Kushlan and Hancock 2005) in reedbeds (Kushlan and Hancock 2005), bushes, trees (Martínez-Vilalta *et al.* 2014) and other plants near water or on islands in sites that are protected from ground predators (Kushlan and Hancock 2005). The species usually nests colonially in single- or mixed-species groups where nests may be less than one metre apart or touching, although they are usually placed more spread out in reedbeds (Kushlan and Hancock 2005). Breeding pairs may also reuse nests from previous years (Kushlan and Hancock 2005). Home ranges for colonies have been investigated in the Neusiedler See population in Austria and where estimated to be in the order of 100-150 km^2 (Nemeth *et al.* 2005). The natal dispersal distance has not been studied in detail, but is expected to be around 700 – 1,200 km (based on Hilbers *et al.* 2016, see Box XXX). The species is a diurnal feeder (Martínez-Vilalta *et al.* 2014) but is most active at dawn and dusk (although in coastal environments its feeding habits are determined by tidal stages) (Kushlan and Hancock 2005), and roosts at night in trees (Brown *et al.* 1982) alongside lakes or rivers or in mangroves, often with other species (Langrand 1990). The species inhabits all kinds of inland and coastal wetlands (Martínez-Vilalta *et al.* 2014) although it is mainly found along the coast in the winter (Snow and Perrins 1998). It frequents river margins, lakes shores, marshes, flood-plains (Martínez-Vilalta *et al.* 2014), oxbows, streams (Snow and Perrins 1998), damp meadows (Kushlan and Hancock 2005), rice-fields, drainage ditches (Martínez-Vilalta *et al.* 2014), aquaculture ponds, reservoirs (Marchant and Higgins 1990, Kushlan and Hancock 2005) and sewage works (Marchant and Higgins 1990, Hockey *et al.* 2005) inland, and the shallows of salt-lakes (Marchant and Higgins 1990), saltpans, mudflats, coastal swamps, (Martínez-Vilalta *et al.* 2014), saltmarshes, seagrass flats, lagoons (Kushlan and Hancock 2005) and estuaries when in coastal locations (Martínez-Vilalta *et al.* 2014). In aquatic habitats its diet consists of fish, amphibians, snakes, aquatic insects and crustaceans although in drier habitats terrestrial insects, lizards, small birds and mammals are more commonly taken (Martínez-Vilalta *et al.* 2014). Foraging distances of up to 4 km from the colony have been recorded in the Neusiedler See colony in Austria (Nemeth *et al.* 2005).
The species is threatened by wetland habitat degradation and loss (Marchant and Higgins 1990, Martínez-Vilalta *et al.* 2014) for example through drainage, grazing, clearing, burning, increased salinity, groundwater extraction and invasion by exotic plants (Marchant and Higgins 1990).

Step 1.2 - Spatial scale of functioning

Breeding population
It is safe to assume on the basis of natal dispersal distance of 700 – 1,200 km (see 1.1) that all Great white Egrets belong to one large continuous European population, consisting of the scattered colonies in Spain and France to the Netherlands in the north and to European Russia in the East.

Passage/winter population
There has been a relatively recent shift in wintering areas from North Africa to closer to the breeding grounds (Lawicki 2014), meaning that the wintering areas may still not be stable, and partially overlap with the breeding areas. It is therefore not recommended to define a FRP and FRR for the non-reproductive population.

Conclusions

Population category MR3 should be applied here, as the Great White Egret is an uncommon species with one population at supranational level. A FRP should be defined for the breeding population at supranational level. Given the complexity and the dynamics of the wintering populations, it is not recommended to define FRVs for the non-reproductive population.

Step 1.3 - Historical perspective: what happened to the species?
The long-term trend (since about 1980) and the short-term trend (since about 2000) are increasing, and as a result the EU population status is 'Secure'. There are no good estimates for the historical trend but the trend likely to be very negative. The species has declined up to the early 20th century with population numbers plummeting, and entire colonies being lost, with a combination of factors is likely to be responsible for the decline: hunting (introduction of firearms), trade in its feathers and large scale wetland drainage (Lawicki 2014). The species went extinct on the Iberian Peninsula and populations in Hungary and Austria virtually disappeared (Lawicki 2014). Given the ongoing recolonization of most of the historical range it is very likely that the species is still recovering from the historic decline.

Step 1.4 - Analysis of distribution and trends
Distribution data and data on numbers are available from Article 12 Bird Directive reporting 2008-2012 (EEA 2015) and Hagemeijer & Blair (1997). The reported data is not accurately reflecting current distribution as a result of many colonisations since 1997.

Breeding population estimates of good or medium quality are available for all EU Member States where the species is breeding (BirdLife 2015). According to these estimates there were 5,200 – 7,700 breeding pairs in the EU in 2007-2012 (BirdLife International 2015). The species also has a (climate) species distribution model, with a 'very good fit' (Huntley et al 2007), however the model is also not accurately reflecting the current climatic range of the species given the recent establishment of the species far outside its predicted climatic range. There are currently no negative trends in population size on Member State level, but the species has declined significantly since historic times.

Conclusions

The long-term trend and the short-term trend are increasing, and the EU population status is 'Secure'. There are currently no negative trends in population size on Member State level, but the species has declined significantly since historic times.

The distribution data is outdated. There are good or medium quality estimates for breeding populations in all EU Member States. The species distribution model is not up to date.

Step 2.1 - FRP assessment
Good densities in the wetland habitats have not been studied, but probably have not been reached yet given that populations in the most important countries are still increasing, such as Ukraine and Hungary. It is also not known which habitat would fall within the climatic requirements of the species. Although the population exceeds its (rule of thumb) MVP already (2500 breeding pairs), for rapidly expanding species recovering from historic decline, the use of operators is recommended until numbers and range of the species have stabilised (the 'wait- and-see' methodology). The FRP for the European breeding population of Great White Egret is therefore preliminary set as much greater than current value.

Conclusions

The Great White Egret is rapidly expanding but is still recovering from a historical population decline, so the FRP for the breeding population is likely to be much higher than the current population:
FRP >> CV.

Step 2.2 - FRR assessment

The current climate models do not capture the distribution of the species adequately, but it is highly likely that the species will continue its rapid range expansion, based on recent colonisations of the species far outside its previously occupied climatic range. The FRR for the European breeding population of Great White Egret is therefore preliminary set as '>> Current range'.

Conclusions

The Great White Egret is rapidly expanding but is still recovering from a historical population decline, so the FRRis likely to be much higher than the current population:
FRR >> CV.

References

BirdLife International (2015). European Red List of Birds. Luxembourg: Office for Official Publications of the European Communities. Including species factsheets.

BirdLife International (2017) IUCN Red List for birds. Downloaded from http://www.birdlife.org on 25/08/2017.

Brown, L.H., Urban, E.K. and Newman, K. 1982. The birds of Africa vol I. Academic Press, London.

Carboneras, C., *et al.* (2016). Handbook of the Birds of the World Alive. Lynx Edicions

EEA (2015). State of Nature in the EU. Luxembourg: Office for Official Publications of the European Union, 2015.

Hafner, H. 2000. Heron nest site conservation. In: Kushlan, J. A.; Hafner, H. (ed.), Heron conservation, pp. 201-217. Academic Press, San Diego

Hagemeijer, W. J., & Blair, M. J. (1997). The EBCC atlas of European breeding birds. T & AD Poyser, London.

Hancock, J. and Kushlan, J. 1984. The herons handbook. Croom Helm, London.

Hockey, P.A.R., Dean, W.R.J. and Ryan, P.G. 2005. Roberts birds of southern Africa. Trustees of the John Voelcker Bird Book Fund, Cape Town, South Africa.

Huntley, B., Green, R. E., Collingham, Y. C., & Willis, S. G. (2007). A climatic atlas of European breeding birds. Barcelona: Lynx Edicions.

Kushlan, J. A. and Hancock, J.A. 2005. The herons. Oxford University Press, Oxford, U.K.

Langrand, O. 1990. Guide to the birds of Madagascar. Yale University Press, New Haven, USA.

Ławicki, Ł. (2014). The Great White Egret in Europe: population increase and range expansion since 1980. British Birds, 107, 8-25.

Marchant, S. and Higgins, P.J. 1990. Handbook of Australian, New Zealand and Antarctic birds, 1: ratites to ducks. Oxford University Press, Melbourne.

Martínez-Vilalta, A., Motis, A. & Kirwan, G.M. (2014). Great White Egret (Ardea alba). In: del Hoyo, J., Elliott, A., Sargatal, J., Christie, D.A. & de Juana, E. (eds.) 2014. Handbook of the Birds of the World Alive. Lynx Edicions, Barcelona. (retrieved from http://www.hbw.com/node/52684 on 15 April 2015).

Nemeth, E., Bossew, P., & Plutzar, C. (2005). A distance-dependent estimation of foraging ranges of neighbouring bird colonies. Ecological Modelling, 182(1), 67-73.

Snow, D.W. and Perrins, C.M. 1998. The Birds of the Western Palearctic vol. 1: Non-Passerines. Oxford University Press, Oxford.

5 Migratory fish

Erwin Winter

5.1 Atlantic salmon (*Salmo salar*) in the Rhine river system

Step 1.1 - Biology of the species
Atlantic salmon (*Salmo salar*) is an anadromous fish species that spawns in the upper reaches of rivers. Eggs are deposited in fast flowing gravel rivers. Juveniles inhabit these rivers for 1-3 years ('parr' stage), after which the fish migrate to sea ('smolt' stage). They spend one to several years at sea to grow to adult size. Important growing areas for the west-European population are the seas around Southern Greenland and around the Faroes islands (Aas *et al.* 2011). The stock in rivers draining into the Baltic have their marine growth stages in the Baltic Sea. Once adult, salmons migrates back to the spawning river where they were born. Natal homing is very strong within salmon, resulting in separate spawning stocks per river system (Aas *et al.* 2011). Salmon live 5-7 years and, depending on different life history traits for different rivers, spawn once (usually in large river systems) or multiple times (usually in small river systems).

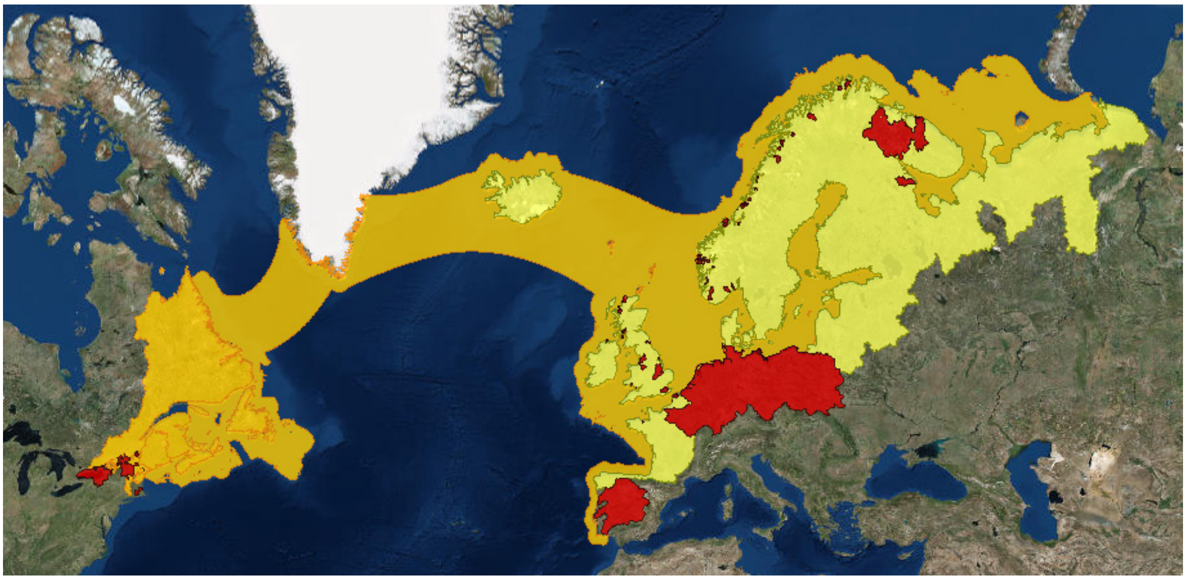


Figure 5.1.1 *Distribution on of Atlantic salmon in freshwater and marine habitats (yellow), river basins where salmon became locally extinct are indicated in red. The marine migratory and feeding areas are indicated in orange (IUCN, 2017)*

Step 1.2 - Spatial scale of functioning (focus on the Rhine river system)
During the marine stages, Atlantic salmon uses the northern part of the Atlantic Ocean and adjacent seas for growth to maturity (Figure 5.1.1). There are three main feeding grounds; west of Greenland and surrounding the Faroes islands and the Baltic. North American Atlantic Salmon that migrate up Canadian and American rivers feed west of Greenland. Salmon that migrates up European, Greenlandic and Russian rivers feed mainly around the Faroes islands but also partially west of Greenland (Figure 5.1.1). Salmon originating from rivers around the Baltic remain in the Baltic Sea during their marine feeding stage. In a few large lakes, e.g. Lake Vättern in Scandinavia, there are land-locked populations that do not migrate to sea (Aas *et al.* 2011). At the marine feeding grounds

many river populations occur mixed together (Figure 5.1.2), but due to their strong homing to natal rivers, populations are separated by river basin with only very limited gene flow or straying between different river populations. Even though individuals have large life time home ranges of several 1000s of kilometres, dispersal between river populations is very limited.

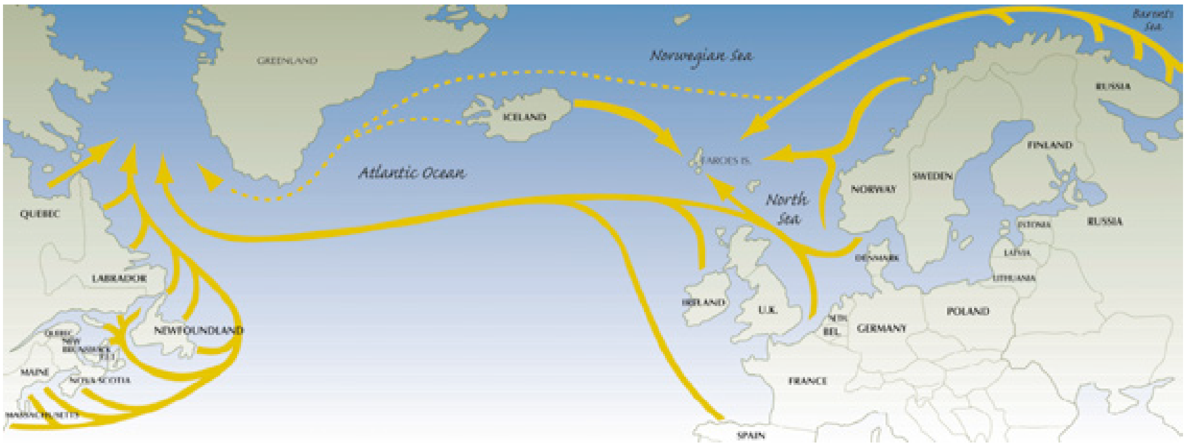


__Figure 5.1.2__ Marine migratory pathways of young salmon (smolts) migrating to the main oceanic feeding grounds. Salmon spawn in the rivers and after a few years moves to one of two feeding grounds in the Atlantic Ocean. Salmon from rivers around the Baltic Sea use the Baltic Sea for growth to mature size

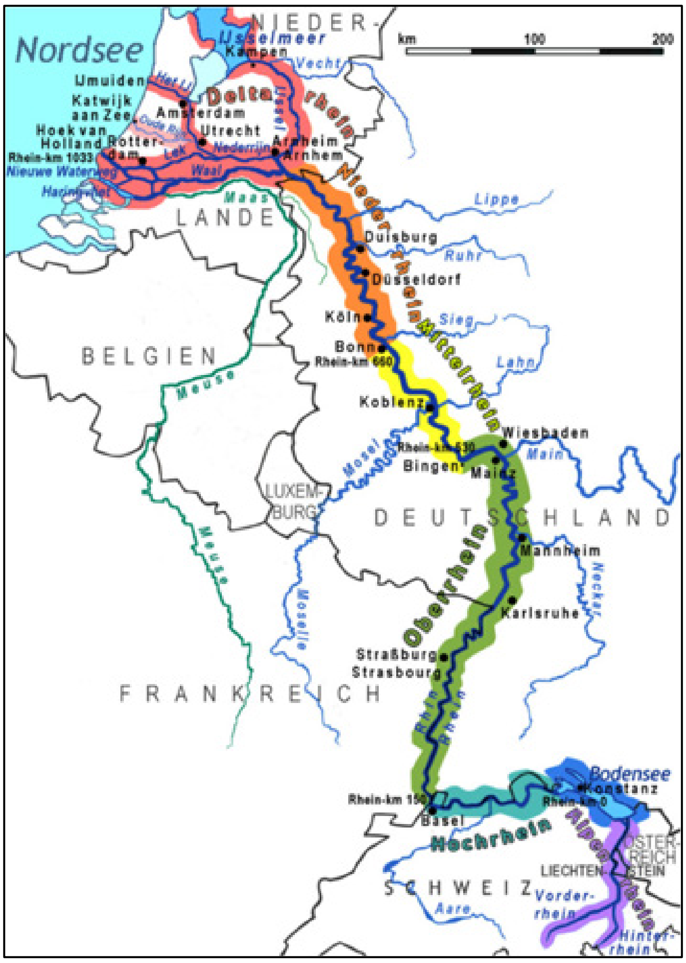


__Figure 5.1.3__ Overview of the river Rhine basin, its main tributaries, and subdivision into different sections of the Rhine (differently coloured)

Salmon populations are clustered at the level of river systems. Salmon passing the Netherlands spawn in the upper reaches of the Rhine. To get there, three routes are available: (i) via Lake IJsselmeer and up the IJssel in the north, or (ii) via the Haringvliet, or (iii) de Nieuwe Waterweg in the south (Figure 5.1.3). The latter is currently the only freely accessible route, whereas the Haringvliet and IJsselmeer can only be accessed by passing discharge sluices. Spawning mainly takes place in the German section of the Rhine, in the Sieg system, the Ahr, the Saynbach, Bruche and Wisper, and to a lesser extent in France, e.g. Mosel (IKSR 2015). The Rhine can be subdivided into different sections in the order from sea to the Alps; 1) Dutch delta of the Rhine (‘*Deltarhein*’), 2) the lower German Rhine (‘*Niederrhein*’), 3) the middle German Rhine (‘*Mittelrhein*’), 4) the upper part of the German Rhine (‘*Oberrhein*’), 5) and the Swiss Rhine sections (‘*Hochrhein*’, ‘*Bodensee*’ and ‘*Alpenrhein*’), see Figure 3. The salmon use the Deltarhein only as a corridor and mainly spawn in the tributaries of the ‘Niederhein’ (Sieg) and ‘Mittelrhein’, and to a lesser extend in tributaries in the ‘Hochrhein’ section. The Swiss Rhine is largely inaccessible to upstream migrating salmon.

Conclusions

For the reproductive populations strong segregation between river basins takes place, where small river basins are at country level MR2 (species with clearly disjunct distributions and one or a few isolated populations at the national level), and larger river systems (e.g. Rhine) act on a supranational scale MR3 (species with individuals with small home ranges and one or a few populations at supranational level). During the non-reproductive marine stages strong mixing between populations occurs at supranational scales MNR3, and partly even outside Europe.

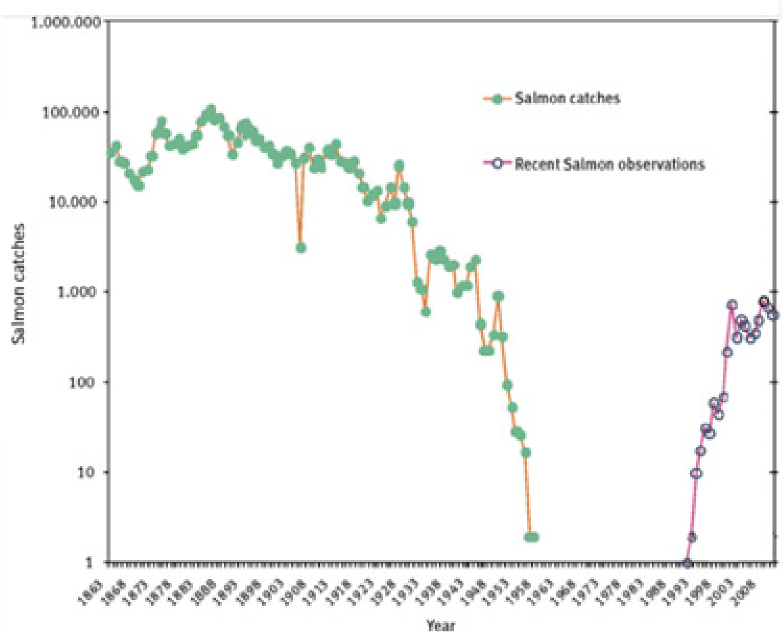


Figure 5.1.4 *Historical catches showing the disappearance of salmon in the Rhine basin in the 1950s and the reappearance since 1993 (adapted from Ingendahl & Beeck, 2011)*

Step 1.3 - Historical perspective: what happened to the species? (focus on the Rhine river system)

The Atlantic salmon stock size has deteriorated greatly in large parts of its distribution area. In some areas/river basins salmon has gone extinct (Figure 5.1.1). The Rhine system at the end of the 1800s was one of the most important salmon rivers in Europe (Ingendahl & Beeck, 2011) and numbers were at least in the 100,000s adult salmon, if not considerably more (Lenders *et al.* 2016, Lenders 2017). Salmon at this time was the most valuable commercial fish species for fresh water fishermen in the Netherlands and Germany. However, salmon declined dramatically around the turn of the century (Figure 5.1.4) despite stocking programs that took place during that time (Lenders 2017), and recent analyses of historical data indicate that this was preceded by earlier declines starting already in the late middle ages (Figure 5.1.5, Lenders 2017) presumably initiated by the large scale construction of numerous small dams with water mills in tributaries of the Rhine blocking spawning migrations and reducing spawning habitat quality (Lenders *et al.* 2016). Finally, like in other adjacent river systems (e.g. Elbe, Thames, Weser), salmon went extinct in the Rhine in the 1950s. Factors driving this extinction were pollution, obstructions in its migration route, deterioration in water quality and overfishing, but also loss of spawning areas (IKSR 2015). As part of a reintroduction program, young salmon of different stages were released in the Rhine from 1988 onwards. Between 1998 and 2009 1-3 million young salmon per year were released (Ingendahl & Beeck, 2011). Subsequently, by the end of the 20th century a more ambitious action project was initiated, to ecologically restore the Rhine and its floodplains, such that fish species such as salmon could return (Rhine Action Plan 2000). In this plan various aspects of the reason for decline are addressed: for example improved accessibility for upstream and downstream migration, improved water quality, improved quality of spawning grounds, as well continuous restocking of young salmon in the next few decennia (Ingendahl & Beeck, 2011, Schneider 2011).

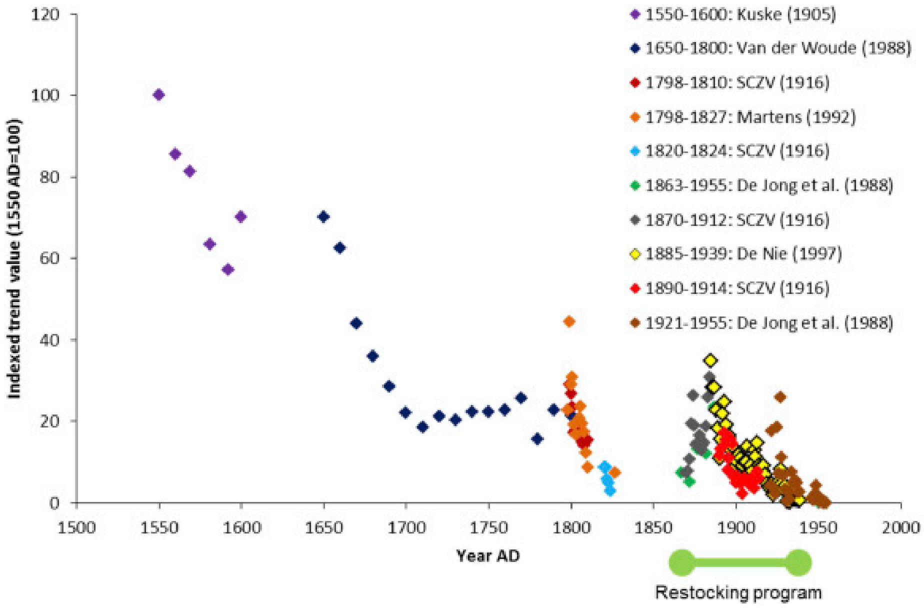


Figure 5.1.5 *Indexed value development of Atlantic salmon (Salmo salar) stocks 1550–1950 in the Lower Rhine. The approximate period of restocking programs, leading to temporarily higher catches, is indicated (from Lenders 2017)*

In 1990 the first adult salmon was caught in upstream parts of the Rhine and in 1994 the first occurrence of natural reproduction was seen (Ingendahl & Beeck, 2011). Around 2007/2008 5-20% of the adult salmon that spawned in the Rhine is believed to originate from natural spawning of returning salmon that were born in the river, while the rest was reintroduced as young salmon. However, the last few years, the percentage salmon of natural spawning origin is believed to be lower (IKSR 2015). The contemporary numbers of returning adult salmon (‘mortality rates from smolt stage to adult stage) are believed to be too low to establish a self-sustaining population (Ingendahl & Beeck, 2011, Schneider 2011)

Step 1.4 - Analysis of distribution and trends

The historical development in population size of salmon has been documented by commercial Rhine catches since 1863 (Figure 4) with 10,000s of salmon caught annually during the early 1900s and at least 100,000s or even more in historical times. A final decline during the first half of the 20th century led to local extinction in the Rhine basin in the 1950s. Recent surveys show an increase in salmon between 1993 and 2003 following reintroduction programs of salmon and ongoing stocking ever since, after which the numbers appear to level off.

A salmon-specific scientific survey is available in all major Dutch river systems: the Rhine (the branches Waal, Nederrijn and IJssel) and the Meuse since 1996 (De Graaf *et al.* 2015). During the migratory period (June, July, October and November) salmon is found at all locations in most years. For the assessment of temporal trends, the most reliable and consistent series is available for the Waal (as this is the main branch of the Rhine river system giving access to upstream spawning habitats). In this river numbers of salmon increased from 1996 up to roughly 2003 and have fluctuated around 0.15 salmon per fyke/24hrs afterwards (Figure 5.1.6).

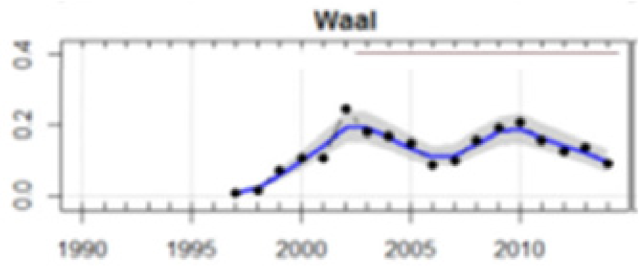
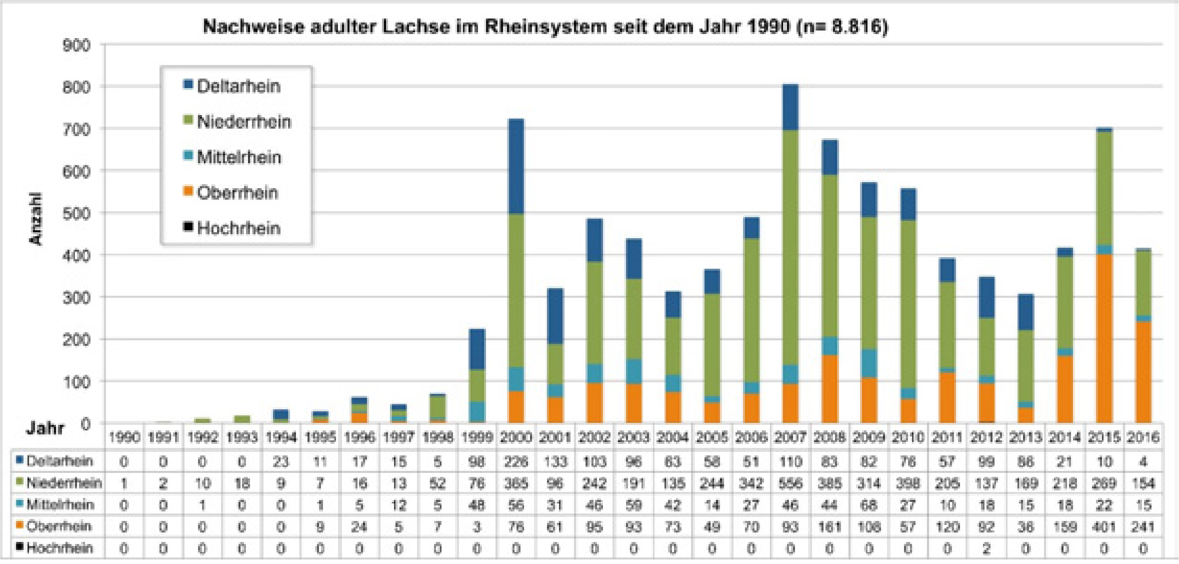


Figure 5.1.6 *Average numbers of salmon caught per 24 hours per fyke, between 1997 and 2014 passing at the salmon survey location in the Waal, part of the Rhine river system, 'Deltarhein' (De Graaf et al. 2015)*



Jahr	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
■ Deltarhein	0	0	0	0	23	11	17	15	5	98	226	133	103	96	63	58	51	110	83	82	76	57	99	86	21	10	4
■ Niederrhein	1	2	10	18	9	7	16	13	52	76	365	96	242	191	135	244	342	556	385	314	398	205	137	169	218	269	154
■ Mittelrhein	0	0	1	0	0	1	5	12	5	48	56	31	46	59	42	14	27	46	44	68	27	10	18	15	18	22	15
■ Oberrhein	0	0	0	0	0	9	24	5	7	3	76	61	95	93	73	49	70	93	161	106	57	120	92	36	159	401	241
■ Hochrhein	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0

Figure 5.1.7 *Numbers of salmon caught within different montoring programs clustered per section of the Rhine, where 'Niederrhein' (encompassing the important tributary Sieg) is the longest and most consistent trend series. The combined numbers of the sections Niederrhein (lower), Mittelrhein (middle) and Oberrhein (upper Rhine) represent minimum estimates for the numbers of spawning adult salmon reaching the spawning grounds, see text step 1.4 for further explanation (IKSR 2017)*

In Germany different montoring programs in fishways along barriers in the different tributaries and mainstream of the Rhine (Koblenz) are available (Figure 5.1.7). The numbers recorded in the Deltarhein are salmon passing thtough the mainstream of the Rhine. These salmon are likely to be 'double counted' when moving into tributaries in the more upstream parts of the Rhine where they are monitored again. The numbers of salmon recorded in the Niederrhein (mainly in the Sieg), Mittelrhein, and Oberrhein are numbers of adult salmon moving through fishways in tributaries that give access to spawning grounds and the numbers combined for these 3 regions are therefore the minimum numbers of returning adults reaching the spawning grounds, i.e. 250-700 on a yearly basis during 2000-2016. There is an increasing trend up to 2000 and no clear gradually fluctuating trend after 2000. In additional some salmon might reach spawning grounds outside monitoring efforts, thus an estimated 500-1000 adult salmon is assessed to be the current spawning population in the Rhine basin. The

Hochrhein and more upstream regions in the Alps are virtually inaccessible to salmon, with only a handfull of salmon recorded in Swiss.

Conclusions

Atlantic salmon declined in many river systems throughout its distribution range during the 20th century, resulting in local extinctions in part of its distribution area, e.g. Rhine, Meuse, Elbe, Seine. In the late 20th century reintroduction programs in different river basins were started. For the Rhine, natural spawning of returning salmon does occur, but stocking remains necessary to this date to sustain the presence of salmon in the Rhine.

Trends suggest an increase in Atlantic salmon in the Rhine after reintroduction began in the late 1980s, which appeared to level off after the early 2000s. Current distribution range is more limited than the historical range due to migration barriers. Minimum numbers of spawning adults are 250-700 per year during 2000-2016 with no clear trend, where total number of salmon that reach the spawning grounds in the Rhine basin is estimated at 500-1,000 adults. Stocking of salmon occurs on a large scale until present, where < 20% of salmon originates from natural spawning. At present the salmon population is not self-sustaining.

Step 2.1 - FRP assessment (Rhine river system)

Atlantic salmon have specific river populations with only very limited exchange between different river systems. During the marine stages however large scale mixing between these populations occur. After this mixed occurrence due to the strong natal homing they migrate back to the rivers where they were born. Therefore FRP should only be assessed for the reproductive populations, setting FRP for the non reproductive stages is neither useful for management purposes nor practical given the vast marine area where numbers should then be estimated. Here we focus on setting FRP for the salmon population spawning in the river Rhine basin which are separated management units that are also more easily assessed. As derived from different monitoring programmes, current population size of adult salmon reaching the spawning grounds in the Rhine basin is at least 250-700 adult salmon per year. Although monitoring programs assess only part of the upstream migrating salmon, it is believed that the majority is recorded. Therefore the actual population size is estimated at least 500-1,000 and presumably not larger than 1,000-2,000. These numbers are achieved due to ongoing stocking of young salmon. If stocking is stopped, numbers will decrease again due to high mortality rates during all the life stages from smolts leaving the spawning areas, to estuaries and open sea, during marine feeding stages and during upstream migration of adult salmon.

Thompson (1991) proposed a generic rule of thumb for MVP of fish species of 1,000 to 10,000 adults. Historically the Rhine harboured a large popualtion of at least 100,000s of salmon. These numbers will not be feasible in the current impacted Rhine system, but a selfsustaning population in the Rhine should at least number 10,000 adults without stocking. Germany and France do not report numbers of salmon in the rivers in the entire country and assess the status at unfavourable (Article 17 2007-2012). The Netherlands report unfavourable status and 500-1,500 salmon passing the Netherlands which is in accordance with the estimation of adults that successfully reach the spanwing grounds, where numbers entering the Netherlands will be higher since some losses occur during upstream migration (Jansen *et al.* 2008). Note that these numbers are substatiated by ongoing stocking and the Rhine salmon population is at present not self-sustaining.

Conclusions

The FRP for the Rhine river system at present is much larger than current value (in 1992 numbers where just picking up after reintroduction programs started in the late 1980s). After 2000 numbers of adults reaching spawning grounds are estimated to 500-2,000 yearly. These numbers are sustained by ongoing stocking and current mortality rates between smolt stages and adult stages are too high to substantiate a self-sustaining population. And therefore population size will decrease if stocking will be stopped. FRP should number at least 10,000 without stocking to have a healthy self-sustaining Rhine population.
FRP >> CV.

Step 4 - FRR assessment (Rhine river system)

The marine distribution range has not changed from historical times to present, even though this might be subject to climate change. The current ditribution range of Atlantic salmon in the Rhine basin is much more restricted than in the historical distribution before the middle ages. Even though the main obstructions in the mainstream and larger tributaries are facilitated with fish passages, many tributaries remain (partly) inaccessible to spawning by adult salmon, e.g. the most upstream parts of the German and all of the Swiss Rhine and the French tributaries, due to the presence of migration barriers. The corridor range in the river in the Netherlands will be largely restored in the coming years, with facilitating upstream migration through Haringvliet and Lake IJsselmeer, which is also reported under Article 17. The historical river range in the upstream spawning areas in Germany and France harboured at least 100,000s adult salmon per year. The current more restriced range can harbour the FRP of at least 10,000 when quality of the spawning grounds is further improved and mortality rates from the smolt stage to adult stages is lowered. France and Germany report ranges for the entire country, of which only a part is situated in the Rhine basin.

Conclusions

FRR has remained the same for the non-reproductive marine stages. For the reproductive stages the current range is more restricted that the historical range that encompassed a larger proportion of the Rhine basin. Since 1992, the distribution range has increased due to facilitating fish passages along barriers in the mainstream and tributaries of the Rhine.
FRR > CV.

References

Aas Ø, Einum S, Klemetsen A, Skurdal J (2011) Atlantic Salmon Ecology. Wiley-Blackwell, Oxford.

De Graaf M, De Boois IJ, Griffioen AB, Van Overzee HMJ, Tien NSH, Tulp I, De Vries P, Deerenberg C (2015). Toestand vis en visserij in de Zoete Rijkswateren: 2014. Deel I: Trends van de visbestanden, vangsten en ecologische kwaliteit ratio’s, IMARES Wageningen UR. IMARES rapport C199/15.

IKSR (2015). De visfauna in de Rijn in 2012/2013. Stand: juli 2015. Internationale Commissie ter Bescherming van de Rijn. www.iksr.org.

Ingendahl D, Beeck P (2011). Critical review on the re-introduction of flagship species. Sciences Eaux & Territories 3.

Jansen HM, Winter HV, Tulp I, Bult T, van Hal R, Bosveld J, Vonk R (2008). Bijvangsten van salmoniden en overige trekvissen vanuit een populatieperspectief. IMARES-rapport C039/08.

Lenders HJR, Chamuleau TPM, Hendriks AJ, Lauwerier RCGM, Leuven RSEW, Verberk WCEP (2016). Historical rise of waterpower initiated the collapse of salmon stocks. Scientific Reports 6: 29269.

Lenders HJR (2017). Fish and fisheries in the Lower Rhine 1550-1950: a historical ecological perspective. J. Environ. Manag. 202 (Part 2), 403e411.

Schneider J (2011). Review of reintroduction of Atlantic salmon (Salmo salar) in tributaries of the Rhine River in the German Federal States of Rhineland-Palatinate and Hesse J. Appl. Ichthyol. 27 (Suppl. 3), 24–32.

5.2 North Sea houting (*Coregonus oxyrinchus*) in the Rhine basin

Step 1.1 - Biology of the species
North Sea houting (*Coregonus oxyrinchus*) is an anadromous fish species endemic to the Wadden Sea and the Dutch Belgian delta of the Rhine, Meuse and Scheldt. Adult houting (36-60 cm) migrate upstream rivers in winter (November-December) to spawn (Jensen *et al.* 2003, Jepsen et al 2012). The eggs are released freely into the water column and stick to gravel and vegetation (Poulsen *et al.* 2012, Muus & Dahlstrøm 1990). It spawns in small rivers and the lower to middle regions of larger rivers that are freely accessible from sea. Houting can return to spawn in the streams several times during their relatively long (10 to 12 yr) lifespan (Borcherding *et al.* 2008, Jepsen *et al.* 2012). Larvae emerge from the eggs in February-March and are roughly 10 mm long (Borcherding *et al.* 2006). Newly hatched fry are carried downstream with the current (Borcherding *et al.* 2006), so their survival depends on being carried into areas of lentic water, such as flooded meadows, reed beds, small impoundments, large river bends, or similar areas which constitute the natural rearing grounds, where the juveniles feed mainly on zooplankton (Borcherding *et al.* 2006). When the fry reach a length of 3 to 4 cm, after about 2 months, they can tolerate marine salinity levels and can migrate/drift out into the sea (Rasmussen 2004). After they migrate to sea, after 1 to 2 (males) or 3 to 4 (females) years, they reach adult size and join the spawning population (Borcherding *et al.* 2006). After becoming sexually mature, the growth rate of houting decreases, and even old (>10 yr) individuals rarely reach lengths of more than 60 cm (Jepsen *et al.* 2003).

Houting is an anadromous species that migrates between marine and freshwater habitats. To what extent individual houting used both freshwater and marine environments in historical times is not well known. For the smaller Danish streams a diadromous migration pattern is clear. In the larger river basins with more extensive estuarine areas, e.g. the Rhine, Meuse and Scheldt delta and the Zuiderzee, it is less clear whether all individuals within the populations migrated to marine habitats for growth. After the estuary Zuiderzee was dammed (in 1932) and a large eutrophic freshwater lake IJsselmeer emerged, houting showed both diadromous and within freshwater movements.

There is some taxonomic controversy on the status of the North Sea houting. Freyhof & Schöter (2005) claimed that the North Sea houting was confined to the Scheldt, Meuse and Rhine basins and that this was a different species than found in the Elbe and Danish Wadden Sea area based on morphological differences (number of gill rakers) between museum houtings from the Scheldt, Meuse and Rhine basins and the houtings from the Vida Aa (which they claimed were in fact C. maraena). This view was adopted by the IUCN and therefore the North Sea houting *C. oxyrhinchus* is listed as globally extinct by the IUCN. Other scientists thereafter dismissed this claim due to the lack of genetic evidence and the strong morphological plasticity that is found in many of the Coregonid species (e.g. Hansen *et al.* 2006, 2008, Jacobsen *et al.* 2012).

Step 1.2 - Spatial scale of functioning (with focus on the Rhine basin)
Adult houting historically inhabited the Dutch, German and Danish Wadden Sea and the Dutch delta. Little is known about the range of distribution in the sea, but it appeared that houting did not migrate far away from the estuaries. Homing of adult houting is very strong, i.e. an adult will spawn in the same fresh water system as where it’s born, and keep going back there over the years. Thus, little mixing between adult populations of different areas in the Wadden Sea and Rhine delata is expected to have occurred.

The only indigenous population of adult houting that has survived lives in the Danish Wadden Sea adjacent to the river Vida. The Vida is the river where this population spawns. In the German Wadden Sea and the Dutch Wadden Sea and delta a few unconnected, reintroduced populations can be found (Figure 5.2.1). The Dutch populations spawn in the river IJssel and the lower reaches of the Rhine-Meuse river system (Rhine basin population). The German populations spawn in the Elbe and a few smaller rivers, e.g. the Treene, between the Elbe and the Danish borders and are considered separate populations (or management units).

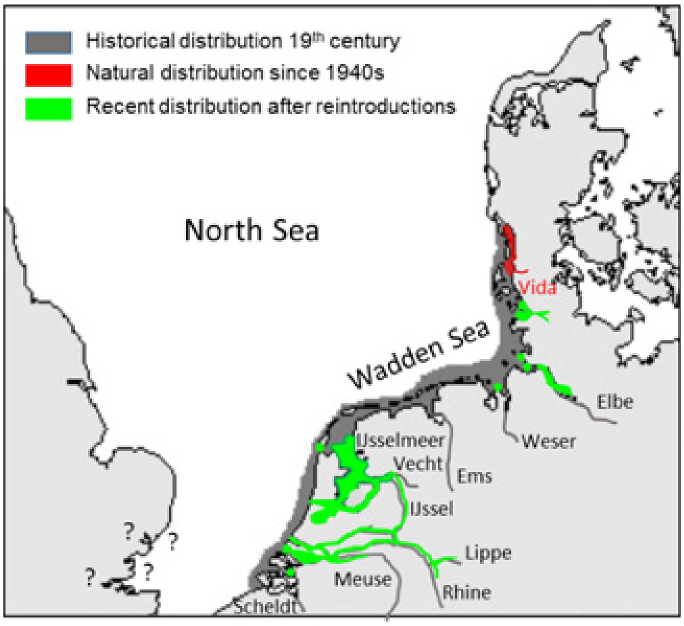


Figure 5.2.1 *Distribution of North Sea houting (modified after Jepsen et al. 2012 and updated for the Rhine basin)*

Since a reintroduction programme in the German Rhine in the 1990s, the IJsselmeer has become an important habitat for adult houting (Borcherding *et al.* 2008). Throughout the year houting of all age classes are found there. Thus, although migration between sea and fresh water remains important for this species, part of the current Rhine population does not use the sea and instead uses the IJsselmeer as main foraging area (Winter *et al.* 2008, Borcherding *et al.* 2008). A large part of the adult IJsselmeer population migrates up the river IJssel in winter for spawning. After spawning in the IJssel it migrates back to the IJsselmeer (Borcherding *et al.* 2014). Only a small part migrates further than the IJssel and moves into the Dutch (lower) parts of German (higher) parts of the Rhine, e.g. the Lippe. Also the lower reaches of the Rhine delta are now used for foraging (Figure 5.2.1).

Conclusions

Houting is a migratory fish species that originally moved between small freshwater streams and the lower and middle regions of larger rivers to estuaries and marine habitats in the vicinity of the river mouths. Most of the reproductive populations can be classified as MR2 (species with clearly disjunct distributions and one or a few isolated (often genetically differentiated) populations at the national level) in the Danish and German Wadden Sea) with only one population (Dutch/German Rhine Basin) that is at supranational scale (MR3) (species with individuals with small home ranges and one or a few populations at supranational level).

Though not much is known on the dispersal during the non-reproductive stages in estuarine and marine habitats, distribution at sea seems restricted from national MNR2 to supranational NMR3.

Step 1.3 - Historical perspective: what happened to the species?

In the 19th century houting was common throughout the Wadden Sea and the adjoining rivers. In the Netherlands it was present in most rivers and the Zuiderzee (van Bemmelen 1866). In the first half of the last century houting declined dramatically in its whole population range. In most river systems houting became extinct. In the Rhine system (Figure 5.2.2) the extinction was caused by the closing of the Zuiderzee (after which it became the fresh water lake IJsselmeer), fisheries, pollution and the disappearance of spawning sites (Poulsen *et al.* 2012). In addition, only rivers with free passage can be used for spawning, as even small weirs and dams block the passage of houting to upriver spawning areas. Fish ladders and small or steep bypass streams do not seem to work for houting, and there is no evidence of spawning upstream of even small obstacles (Jensen *et al.* 2003, Jepsen et al 2012).

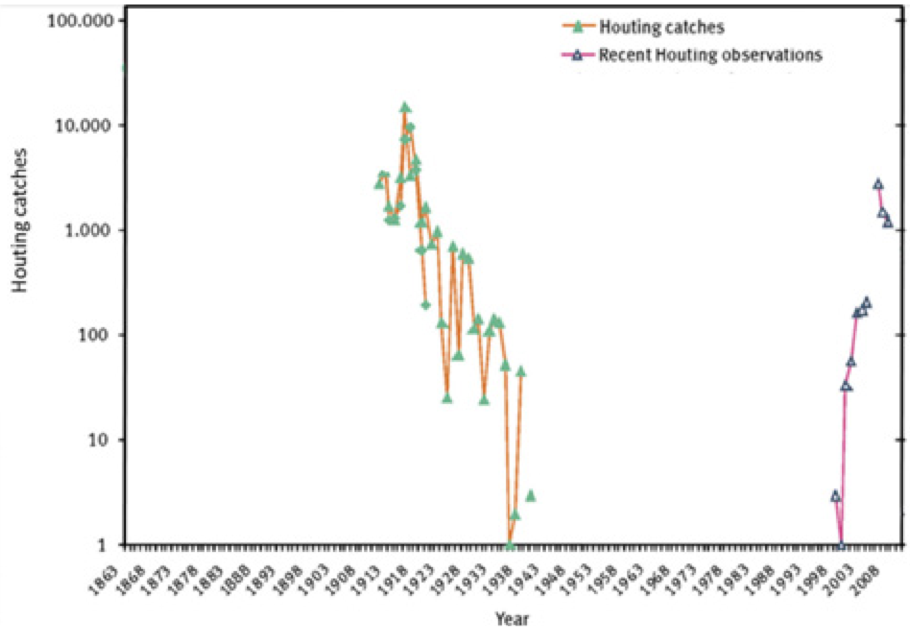


Figure 5.2.2 *Historical catches showing the disappearance of houting in the Rhine basin in the 1930s and the reappearance after the introduction program started in 1990s (modified from Ingendahl & Beeck, 2011)*

Only one indigenous population of houting remains in the world, located in the Danish River Vidaa (Jepsen *et al.* 2012). Fish from the Vidaa have been used as broodstock for reintroduction of houting in other rivers (Borcherding *et al.* 2010). Between 1987 and 1992 houting was reintroduced in the Danish Wadden Sea (Jepsen *et al.* 2012) and since the nineties of the last century a large-scale reintroduction programme was executed in the German Rhine and the adjacent river Lippe (Borcherding *et al.* 2010). This German programme was very successful. Nowadays, next to the Vidaa/Danish Wadden Sea population, small reintroduced populations are found in both German and Dutch parts of the Wadden Sea and the adjacent fresh water areas (Figure 5.2.1).

Step 1.4 - Analysis of distribution and trends

For houting residing in Dutch waters, data on temporal trends is scarce. There are no surveys that cover the upstream migration period (November-December). There is a dataset (up to 2008) that contains the houting catches of commercial fyke fishermen during the non-migration period (in May-September). These data show that houting is only found in significant number in the northern part of the IJsselmeer near the Afsluitdijk (=dam between Wadden Sea and IJsselmeer). Relatively small numbers of houting were caught in the southern part of the IJsselmeer, the Rhine, Meuse and their delta and no upward (or downward) trend can be detected in these areas (reference). Near the Afsluitdijk, houting survey catches increased from 2000 up to 2008.

In the absence of long-term data series, shorter lasting monitoring programs, project studies (including telemetry studies in Lake IJsselmeer and the lower branches of the Rhine) and anecdotal catch information from professional fishermen can be used to assess the status of houting in the Rhine basin (Winter *et al.* 2008, Bosveld 2008). Houting maintained its presence in the basin, in the IJsselmeer/IJssel, after the stocking ceased in 2006, in which year >95% of young houting originated from natural spawning. Also, with relatively little catching effort in Lake IJsselmeer 50 adult houting were caught by seining (Bosveld 2008).

Conclusions

The North Sea houting that is endemic to the Wadden Sea and Dutch delta became nearly extinct in the early 20th century. Only one small population remained in the small Danish stream Vidaa. Since the 1990s reintroduction programs were started in several Danish and German streams and rivers and in the German part of the Rhine Basin. Since then a self-sustaining houting population in the lower reaches of the Rhine basin (in particular IJsselmeer/IJssel) was re-established and stocking ceased after 2006.

The total population size in the Rhine basin is unknown. Also trends of houting numbers are incomplete. Houting definitely increased in one part of the Rhine basin (Ijsselmeer/Ijssel) since 1996 when the reintroduction program started. Presence in this part has remained stable after stocking ceased in 2006 and a viable population seems to be present here. In other parts of the Rhine basin, no evidence for an improved population size since 1996 is present although it is known that to some degree of mixing between these areas occurs.

Step 2.1 - FRP assessment

Current population size of houting in the Rhine basin is unknown. However, houting maintained its presence in one part of the basin (the IJsselmeer/IJssel), after re-stocking ceased in 2006. Thus a viable population seems to be present there of at least 1,000s of adults, most likely more. In other parts of the Rhine basin the number of houting caught in the surveys remains low. To what degree houting in these basin parts are part of the same population is unknown, but at least some mixing in the lower Rhine delta occur, as demonstrated with telemetry (Bosveld 2008).

Thompson (1991) states a generic rule of thumb for MVP of fish species of 1,000 to 10,0000 adults. Denmark estimates the current population size at 4,000-6,000 adult fish with a stable trend and rates this as unfavourable (Article 17 2007-2012), which is in accordance with the numbers for an MVP as proposed by Thompson plus a safety marge. Germany does not report values for houting but rates its status as unfavourable. The Netherlands do not report houting so far, since it was considered extinct in 1992, and its reappearance since the reintroduction started in the German Rhine since the 1990s has not been reported under Article 17 yet.

Conclusions

Houting was extinct in the Rhine basin in 1992 but currently present as a self-sustaining population in the IJsselmeer/IJssel. Thus, FRP >> CV. The current FRP is unknown, but is likely to be at least higher than the minimum viable population size.

To what degree houting in other parts of the Rhine basin are part of the same population is unknown. No evidence for increased numbers in these areas is present.

Step 2.2 - FRR assessment

The historical distribution range of houting in the Rhine basin is not well known. The current range in the Rhine basin encompasses the different Rhine branches in the Netherlands, marine areas just downstream from the river mouths in the Wadden Sea and the Haringvliet and upstream to the Lippe and lower reach of the German Rhine (Figure 5.2.2). This is believed comparable to what its range in the River basin was in historical times. There are no indications so far however that the Rhine population has extended its range to formerly occupied ranges in the lower reaches of the Scheldt. The current ranges for Denmark and Germany are reported under Article 17, however the current distribution in the Netherlands (as indicated in Figure 5.2.2) has not been reported yet.

Conclusions

The historical range of houting in the Rhine basin is not well known. The current range in the Rhine basin encompasses the different Rhine branches in the Netherlands, marine areas just downstream from the river mouths in the Wadden Sea and the Haringvliet and upstream to the Lippe and lower reach of the German Rhine. This is believed comparable to what its range in the River basin was in historical times. FRR = CV.

References

Bosveld J (2008). The status and degree of rehabilitation of populations of Houting (Coregonus oxyrinchus) and Twaite shad (Alosa fallax) in the Netherlands. Report Environmental Science no. 325. Radboud University, Nijmegen

Borcherding J, Scharbert A, Urbatzka R (2006) Timing of downstream migration and food uptake of juvenile North Sea houting stocked in the Lower Rhine and the Lippe (Germany). J Fish Biol 68: 1271−1286

Borcherding J, Pickhardt C, Winter HV, Becker JS (2008) Migration history of North Sea houting (Coregonus oxyrinchus L.) caught in Lake IJsselmeer (The Netherlands) inferred from scale transects of Sr-88 : Ca-44 ratios. Aquatic Sciences 70:47-56

Borcherding J, Heynen M, Jäger-Kleinicke T, Winter HV, Eckman R (2010). Re-establishment of the North Sea houting in the River Rhine. Fisheries Management and Ecology 17: 291-293.

Borcherding J, Breukelaar AW, Winter HV, König U (2014) Spawning migration and larval drift of anadromous north sea houting (Coregonus oxyrinchus) in the river IJssel, the Netherlands. Ecology of Freshwater Fish 23: 161–170.

Hansen M, Nielsen EE, Mensberg KLD (2006). Underwater but not out of sight: genetic monitoring of effective population size in the endangered North Sea Houting (Coregonus oxyrhynchus). Canadian Journal of Fisheries and Aquatic Sciences, 63 (4): 780-787.

Hansen MM, Fraser DJ, Als TD, Mensberg KLD (2008). Reproductive isolation, evolutionary distinctiveness and setting conservation priorities: The case of a European lake whitefish and the endangered North Sea houting (Coregonus spp.). BMC Evolutionary Biology 8: 137

Ingendahl D, Beeck P (2011). Critical review on the re-introduction of flagship species. Sciences Eaux & Territories 3.

Jacobsen MW, Hansen MM, Orlando L, Bekkevold D, Bernatchez L, Willerslev E, Gilbert MTP (2012). Mitogenome sequencing reveals shallow evolutionary histories and recent divergence time between morphologically and ecologically distinct European whitefish (Coregonus spp.). Molecular Ecology 21:2727–2742.

Jensen AR, Nielsen HT, Ejbye-Ernst M (2003) National manage ment plan for the houting. Ministry of the Environment, Forest and Nature Agency, County of Southern Jutland and County of Ribe, Ribe

Jepsen N., Deacon M., Koed A. (2012) Decline of the North Sea houting: protective measures for an endangered anadromous fish. Endangered Species Research Vol. 16: 77–84

Muus BJ, Dahlstrøm P (1990) Europas ferskvandsfisk. Gads Forlag, Copenhagen

Poulsen SB, Jensen LF, Schulz C, Deacon M, Meyer KE, Jager-Kleinicke T, Schwarten H, Svendsen JC (2012) Ontogenetic differentiation of swimming performance and behaviour in relation to habitat availability in the endangered North Sea houting (Coregonus oxyrinchus). Aquatic Living Resources 25:241-249

Rasmussen PC (2004) Opvækstområder for snæbel i Vidå og Ribe Å. County of Sønderjylland, County of Ribe, and Forest and Nature Agency, Ribe

Thompson GG (1991). Determining minimum viable populations under the endangered species act. NOAA Technical Memorandum NMFS F/NWC-198.

Winter HV, de Leeuw JJ, Bosveld J (2008) Houting in het IJsselmeergebied. Een uitgestorven vis terug? - IMARES rapport nr. C084/08.

5.3 River lamprey (*Lampetra fluviatilis*) in the Scheldt-Meuse-Rhine basin

Step 1.1 - Biology of the species
River lamprey (*Lampetra fluviatilis*) is a species of jawless fish (*Agnatha* consisting of both present *(Cyclostomata)* and extinct *(Conodonts* and *Ostracoderms*) species). Adult river lamprey are 30-45 cm long and live as ectoparasites in coastal waters and estuaries. They feed on a variety of fish, such as herring, sprat and flounder. They feed by rasping away large amounts of flesh of the back of a fish (Maitland 2003). After one to two years, in October to December, river lamprey migrate up medium/large rivers. During winter and into early spring they migrate upstream during the night and hide under stones and vegetation during the day. Spawning takes place in March-April, in clean rivers and streams with gravel and small stones and flowing water (Morris & Maitland 1987). Several adults create a nest (a depression up to 40 cm in diameter) in which the females spawn over the course of several days. The eggs stick to the sand and the nest substrate. All adult lampreys die after spawning. After 15-30 days the larvae hatch and drift downstream to suitable silt beds (Maitland 2003). Larvae spend several years buried in these silt beds and feed as filterfeeder, mainly on micro-organisms. After metamorphosing in spring (at 9-12 cm and 3-5 years) the young adults migrate downstream to the estuaries and coastal waters (Maitland 2003). They spend their time in areas near the river from which they migrated.

Step 1.2 - Spatial scale of functioning (with focus on Scheldt-Meuse-Rhine basin)
River lamprey has been found mainly in western Europe, from southern Norway to Portugal and also in the western Mediterranean (where it is supposed to be extinct now, Figure 5.3.1). The species is mostly anadromous but there are some land-locked, non-migratory populations in Finland, Russia and Scotland (Maitland 2003).

Figure 5.3.1 *Spatial distribution of river lamprey during their freshwater life stages. Orange is current distribution, red are river basins where river lamprey is considered extinct. In addition to this, river lamprey occurs in marine coastal waters and estuaries adjacent to the freshwater distribution (IUCN, 2017)*

River lampreys spawn in small streams and the lower to middle reaches of larger rivers. The larvae drift from the spawning sites to silt banks in flowing water. After this initial growth phase in freshwater a metamorphosis takes place and they migrate to sea. Dispersal at sea is not well known, but data from surveys at sea suggest that they remain in coastal waters (Figure 5.3.2).

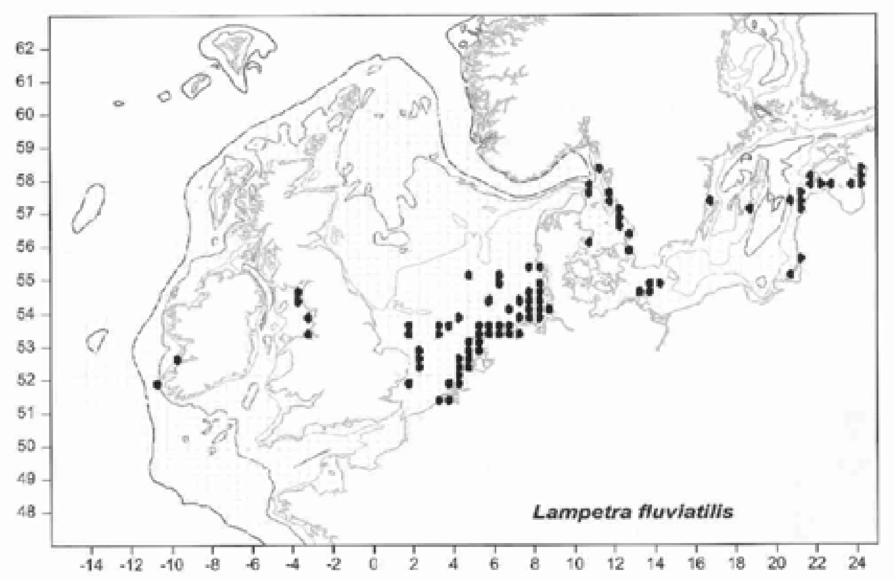


Figure 5.3.2 *Occurrence of river lamprey in all conducted surveys at sea or coastal areas from the ICES database (Heessen et al. 2016)*

The Rhine, Meuse and Scheldt rivers discharge into sea through a large delta. This delta has been strongly modified by humans through poldering, building dams and dikes and diverting river courses. Historically these river basins that encompass Belgium, the Netherlands, France and Germany, were used by river lamprey up to at least the middle reaches and tributairies. Little is known about the current spawning areas in the Scheldt, Meuse and Rhine basin although an increasing number of sites has been discovered in the past years (Winter *et al.* 2013, Spikmans *et al.* 2016). Individuals can move over distances of several hundreds of kilometers as derived from catches in e.g. the fishway at Ifezheim in the German Rhine.

Contrary to other anadromous species, for river lamprey there is no evidence for site fidelity and homing to their natal river for spawning (Waldman *et al.* 2008). Instead they exhibit regional panmixia, i.e. random mating between individuals from the same coastal region.

Conclusions

The spatial scale necessary for a complete life cycle of the River lamprey is in the order of magnitude of several hundreds of kilometer. There is no natal homing and therefore regional mixing between reproductive populations in adjacent rivers occurs. Reproductive and non-reproductive populations overlap and together can be classified as MR3 (species with individuals with small home ranges and one or a few populations at supranational level).

Step 1.3 - Historical perspective: what happened to the species?

River lamprey was widely distributed in Northern Europe and historically penetrated deep into central Europe. It has however declined over the last hundred years due to pollution, man-made obstacles in its migratory route and intensive fishing (Maitland 2003). It has disappeared from many rivers. In the Rhine system is has never disappeared although abundance has dropped strongly in the 20th century. Historical data on catches in the1840s, when there was a fisheries in the Dutch Rhine branches with small cubs on river lamprey for the use of bait within other fisheries e.g. cod fisheries, suggest that river lamprey was a highly abundant migratory species in the rivers Rhine and Meuse in historical times (van den Ende, 1847). In the Meuse at Lith, Lanzing (1959) reported estimated commercial catches of 95,000-200,000 adults yearly between 1953-1957 at this one site. Anecdotal data from professional fishermen suggest that in the 1960s and 1970s river lamprey was at its lowest abundance in the Rhine basin and that population levels have increased in the 1980s and 1990s, but the species had never disappeared as was the case for other diadromous species in the Rhine and Meuse catchments (such as Allis shad, European sturgeon, houting and Atlantic salmon). Reliable trend data however are not available due to the elusive life cycle of the lamprey including upstream migration of adults taking place in late autumn-early winter (migration peak mostly in December) when few if any fish surveys or monitoring take place. During their freshwater phase, the larvae live buried in soft sediments which requires a targeted survey or monitoring to detect them. They are hardly ever caught within other fish or macro-fauna monitoring programmes. The developmental stage that migrates from freshwater to sea is narrow and eel-like bodied with only 12-14 cm in length and therefore only detectable in fishing gears with very small mesh-size. Thus, the presence of river lamprey can easily remain undetected, even when present in relatively large abundance, in the absence of monitoring programs especially targeting river lamprey.

Step 1.4 - Analysis of distribution and trends

Little is known about the trend in distribution and population size of river lamprey due to its elusive life cycle (see step 1.3). Numbers of river lamprey caught in fyke net monitoring programs that only reflect the start of the river lamprey migration (the majority of the migration period remains outside this monitoring period), show that year to year fluctuations can vary orders of magnitude; 100s to 1,000s of river lamprey caught per year during 1993-2008 (Winter *et al.* 2014). This is more likely related to yearly variation in timing of migration (i.e. early start versus late start of migration) than to fluctuations in population size. After 2009 the eel fisheries in the Rhine and Meuse branches was closed and the fyke net monitoring program was altered severely with much less fishing effort remaining within the monitoring program.

Conclusions

Strong decline occurred during the 20th century resulting in extinctions in some river basins. In the Rhine population levels were most probably at their lowest during the 1960s and 1970s, followed by an increase in the 1980s and 1990s. Numbers appear to level off during the 21th century, but no reliable trend estimates are available at present due to the elusive life cycle of the river lamprey and timing of migration.

Numbers caught in programs that are in fact unsuitable for the species (in terms of timing and net efficiency), strongly vary from year to year, which is more likely to be related to yearly variation in the start of migration than variation in actual population size.

Step 2.1 - FRP assessment

Total numbers of 100s to 1,000s caught within fyke nets monitoring programs during 1993-2008 (Winter et al 2014). In short lasting studies, sometimes relatively large numbers of river lamprey are caught at sites within the Dutch delta, e.g. over 4,000 river lamprey during 4 nights in March 2006 in a fishway monitoring in the Nederrijn/Lek branch of the Rhine (Winter 2010). In a mark-recapture study it was estimated that 1,000s-10,000s river lamprey were present during the migration period in the Wadden Sea at Kornwerderzand sluices accessing Lake IJsselmeer (Griffioen *et al.* 2014). In the German River Ems 20,000-40,000 river lampreys were observed yearly migrating upstream along the most downstream situated weir in 2011 and 2012 (Scholle 2012). Based on numbers caught within programs that are unsuitable for this species in timing, and considering that catching efficiency of the fyke nets is usually low, the population numbers of river lamprey entering the Dutch delta per migration season are at least in the 10,000s and most probably much higher than that.

Thompson (1991) proposed a generic rule of thumb for MVP of fish species of 1,000 to 10,000 adults. Belgium reports a very wide range of 4,310 – 4,310,000 individuals, underlining the uncertainties for this species as mentioned above, with an increasing trend and rates this as unfavourable (Article 17 2007-2012). Germany does not report numbers, but assesses its status as favourable for all the German rivers. The Netherlands reports a range of 10,000–100,000 individuals with an unknown trend and assess its status as favourable. For the entire Scheldt-Meuse-Rhine basin a FRV of at least 10,000-100,000 would be appropriate based on the indications for its abundance in the past and present, though good estimates are currently not available.

Conclusions

Total numbers caught in monitoring programs that are not well-suited to detect river lamprey migration indicate that current population levels entering the Scheldt-Meuse Rhine delta are at least in the 10,000s, probably much higher, and therefore above viable population level, especially when taking into account that there is also regional mixing between populations spawning in adjacent rivers.

FRP (Scheldt-Meuse-Rhine basin)>>10,000 is proposed for river lamprey.

Step 2.2 - FRR assessment

River lamprey is present in all the mainstream branches of the Rhine and Meuse within the Netherlands and in the lower branches of the Scheldt in Belgium and the Rhine up to Ifezheim (Figure 5.3.3). In addition to this (and relevant for the Scheldt-Meuse-Rhine basin population) they are also found in some smaller tributaries, e.g. the Niers and Roer in the Meuse basin and Lippe and Sieg in the Rhine basin. In historical times much more small streams and tributaries contained river lamprey, but due to migration barriers, e.g. sluices and weirs, these were no longer accessible to river lamprey. Since 1992, due to the construction of fishways the range of river lamprey has most probably extended.

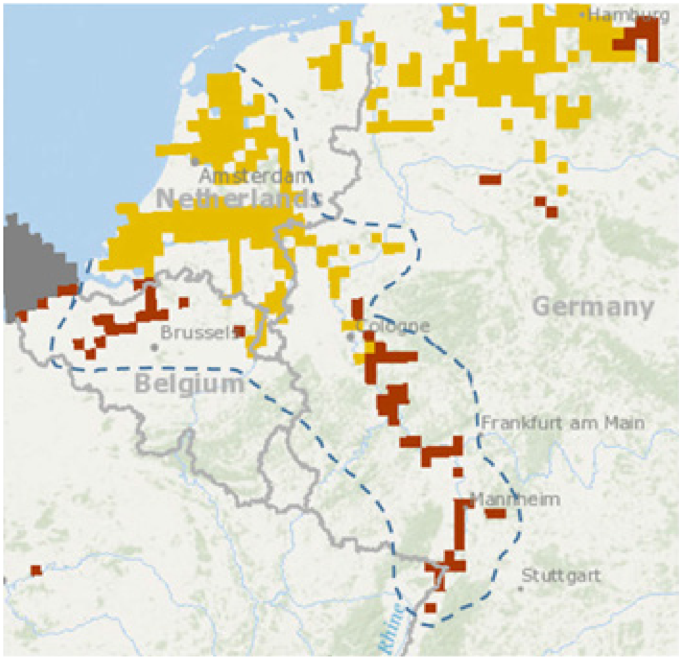


Figure 5.3.3 *Range of occurence of the river lamprey as reported under Article 17 by Belgium, the Netherlands and Germany. The range assessments that lie within the Scheldt-Meuse-Rhine basin are indicated by the blue dotted line*

Conclusions

Although the current range is sufficient to harbour the minimum required population size, it is more restricted than the historical range, mainly due to migration barriers prohibiting migration into small streams and tributaries. Distribution range in the larger rivers, estuaries and at sea are similar to historical ranges (see FRP above).

FRR (Scheldt-Meuse-Rhine basin) > CV.

References

Gaudron SM, Lucas MC (2006). First evidence of attraction of adult river lamprey in the migratory phase to larval odour. Journal of Fish Biology 68, 640-644

Griffioen AB, Winter HV (2014). Inschatting van het aanbod diadrome vis bij Kornwerderzand.IMARES-report C069/14.

Griffioen AB, Winter HV (2014). Merk-terugvangst experiment rivierprik (Lampetra fluviatilis) bij Kornwerderzand. IMARES-report C044/14.

Griffioen AB, Winter HV (2014). Het voorkomen van diadrome vis in de spuikom van Kornwerderzand 2001-2012 en de relatie met spuidebieten. IMARES-report C036/14.

Lanzing WJR (1959). Studies on the River Lamprey, Lampetra Fluviatilis, during its anadromous migration. PhD-thesis. Neerlandia, Utrecht. 82 pp.

Maitland PS (2003). Ecology of the River, Brook and Sea Lamprey.Conserving Natura 2000 Rivers Ecology Series No. 5. English Nature, Peterborough.

Morris KH, Maitland PS (1987). A trap for catching adult lampreys (Petromyzonidae) in running water. Journal of Fish Biology 31, 513–516.c

Scholle J, Kopetsch D, Rückert P, Bildstein T, Meyerdirks J (2012) Herstellung der durchgängigkeit für fische und rundmäuler in den vorranggewässern der internationalen flusgebietseinheit Ems. In. BIOCONSULT

Spikmans F, de Bruin A, Kranenbarg J (2016). Verkennende studie naar voorkomen larven rivier- en zeeprik in de Maas. Stichting RAVON rapport 2015.032, Nijmegen.

Thompson GG (1991). Determining minimum viable populations under the endangered species act. NOAA Technical Memorandum NMFS F/NWC-198.

Tuunainen P, Ikonen E, Auvinen H (1980). Lamprey and lamprey fisheries in Finland. Canadian Journal of Fisheries and Aquatic Sciences 37, 1953–1959.

Van den Ende WP (1847). Verslag der werkzaamheden Vereeniging tot Bevordering der Inlandsche Ichtyologie: eerste en tweede deel. An. Nijhoff, Arnhem.

Winter HV (2007). A fisheye view on fishways. PhD Thesis, Wageningen University, The Netherlands.

Winter HV, Griffioen AB, van Keeken OA (2014). De Vismigratierivier: Bronnenonderzoek naar gedrag van vis rond zoet-zout overgangen. IMARES-report C035/14.

Winter HV, Griffioen AB, van Keeken OA, Schollema PP (2013). Telemetry study on migration of river lamprey and silver eel in the Hunze and Aa catchment basin. IMARES-report C012/13.

Winter HV, Griffioen AB (2007). Verspreiding van rivierprik-larven in het Drentsche Aa stroomgebied. IMARES-rapport C015/07.

Winter HV (2010). Evaluatie van de vistrappen in de Nederrijn-Lek. IMARES Rapport C064/10.

6 Invertebrates

6.1 Large copper (*Lycaena dispar*) in the Netherlands

Henk Siepel & Chris van Swaaij

Step 1.1 - Biology of the species

The last remnant European population of the subspecies *Lycaena dispar batava* occurs in the Northwest of the province Overijssel and the Southeast of the province Friesland (Figure 6.1.1) where it was discovered in 1915. The nominate subspecies (*L. dispar dispar*) known from the UK has gone extinct already in 1851. Two other subspecies occur elsewhere in Europe: *L. dispar rutila* in e.g. Northern Germany, Poland, Czech Republic, Slowakia and *L. dispar carueli* in Belgium, France and Southwest Germany. The latter subspecies are clearly smaller and have two generations a year while *L. dispar dispar* and *L. dispar batava* only have one.

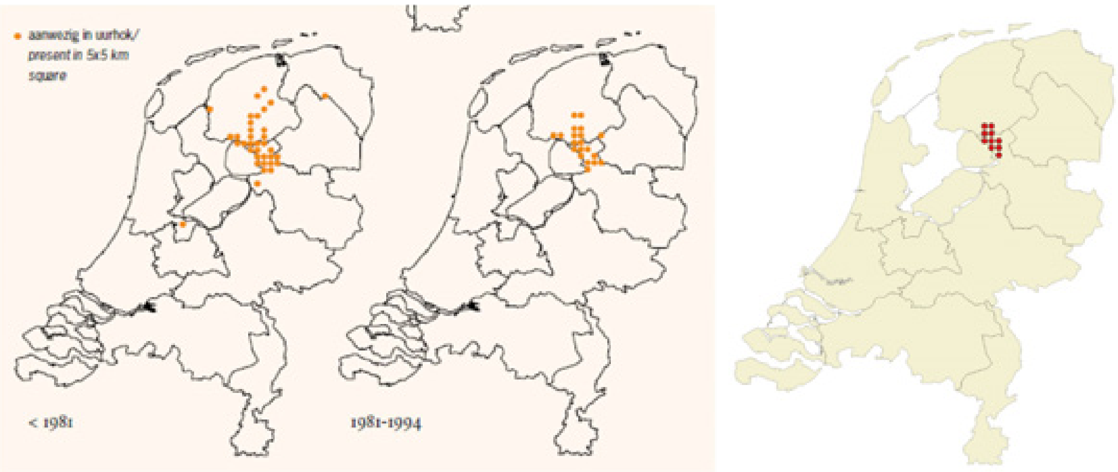


Figure 6.1.1 *Distribution of the Large Copper (Lycaena dispar batava) in the Netherlands in the periods <1981, 1981-1994 (from Bos et al. 2006) and 2010-2017*

The span width of this butterfly is 27-32 mm. It has one generation per year and the caterpillars feed mainly on leaves of the Great water dock (*Rumex hydrolapathum*) found in lowland marshes with a reed and sedge-vegetation. Potential egg production is comparatively large (about 150 on average in natural conditions with a maximum of 700 eggs/female in cultures) and females have a selective oviposition. Males are territorial. Egg production of females scores highest on hostplants kept at a soil pH of 5.5 at which the plants contain the highest amount of nitrogen and a favourable N/P ratio of 9.2/ Acid stress of hostplants decreases egg production (Bink 1986). Caterpillars hibernate in the 3rd instar at the base of the hostplant. Adults are active from the end of June till the end of August. Development times are: eggs 8±3 days, caterpillars 320±20 days (including hibernation) and pupae 18±6 days. This subspecies occurs in a much lower density (about 4 individuals per ha) than the other subspecies (Bink 1992; 2013).

Step 1.2 - Spatial scale of functioning

Large coppers have a very modest dispersal capacity of usually at most 5 km, though in some years it can be higher. Surplus production in a core area results in more dispersal as males behave territorial and females have a selective oviposition. Dispersal will be low to absent at low densities.

Conclusions

This species could be listed under population category S5 (small, mobile animal species with year-to-year variation in occurrence of suitable habitat) as this butterfly has a modest mobility and dispersal capacity, living in discrete subpopulations with a small exchange resulting in one effective population at the national level.

Step 1.3 - Historical perspective: what happened to the species?
From early Medieval times to about a century ago much of the lower peatlands in the Netherlands have been used for peat digging (fuel) and were cultivated, which probably resulted in a major contraction of the subspecies' range before data were sampled. Although the area of the subspecies was already much smaller for the period we have data, it is clear that this area is even more contracted during the last century (Figure 6.1.1) resulting at this moment in one core population and two smaller satellites with a total range of 4 10 x 10 km squares (400 km^2).

Step 1.4 - Analysis of distribution and trends
The last core population occurs in the Natura 2000 site Weerribben (Northwest of the province Overijssel); the two satellite populations are in the Natura 2000 site Rottige Meenthe (Southeast of the province Friesland, at 4 km distance from the core population) and in the Natura 2000 site De Wieden (Northwest of the province of Overijssel, at a distance of ca. 8 km from the core population). The estimated current size of the core population is 250-700 individuals while both satellites are much smaller: Rottige Meenthe 20-100 and De Wieden <10 (Groenendijk & Van Swaay 2005, Van Swaay *et al.* 2017). In Table 6.1.1 location, year of extinction and distance from the core are given of all known Dutch populations in the past century.

Table 6.1.1 *Locations with populations of the Large copper, with their surface area, year of extinction (last sighting) of the population, the distance from the core area and current population size.*Given is the total area of the reserves, not the area of suitable habitat, which can be a fraction of this total area*

Name reserve	Surface area* (ha)	Year of last sighting	Distance from core area (km)	Current population size
Weerribben	3160	--	core area	250-700
Rottige Meenthe	1304	--	4.0	20-100
De Wieden	5000	--	10.6	<10
Bancopolder	21	1996	19.2	--
Oosterschar	345	1992	13.6	--
De Deelen	1157	1985	25.1	--
Makkummerwaard	900	1982	62.6	--
Lindevallei	535	1971	11.3	--

Although the core population with an estimate of 250-700 individuals is not directly endangered, the long-term viability of the species is not guaranteed. From Table 6.1.1 it is obvious that the more remote or the smaller an area is, the sooner the population went extinct. Despite the large area of De Wieden, about 40-60% of this site is forested and unsuitable habitat. Given the small remnant population in that area one may expect local extinction in the coming years. Recolonization depends on the surplus of the population in the core area, which fluctuates. Given the distances to the former populated areas, it is very unlikely that these will become repopulated spontaneously (except for the Wieden area, but see above). Earlier (re)introduction attempts have failed in 1928 and 1932 (Naardermeer) and 1932-1940 (Boornbergum and Eernewoude) (Bink, 1992), probably due to lack of knowledge of exact conditions required by the species and due to the combination of a low natural density (territoriality) and specific habitat requirements. The species is a limited disperser and when suitable areas are in the neighbourhood (distance < 5 km), one may expect recolonization. Repopulating more remote locations is only recommended, when connectivity to other sites or the core population is foreseen in the near future.

In summary: Numbers in the core population fluctuate, but not to a level that a surplus of the local population can be expected. Given its high fecundity an increase of the population can be expected when habitat conditions become more favourable. Although we see at the moment a positive trend in numbers in the core population, we also witness a series of extinctions of satellite populations, which make the (sub)species vulnerable to extinction.

In order to get a more sustainable population again, acidification of the fens around the core population must be reversed. Acidification of the fen vegetation can form a threat to the species survival (see step 1.1), as do other types of loss of habitat (woodland encroachment). Restoration of sites to a neutral to slightly acid (pH 5.5) reed land with enough host plants is recommended. Ongoing succession, without newly formed open ditches is also a major problem (Van Swaay & Bink 2000). Only when the core population increases to larger densities, colonization of more remote (5 km) areas can be expected.

Conclusions

Although the Large copper never had a very extended range in historical times, it managed to survive in a more or less sustainable population. During the past half century however, many satellite populations went extinct. Recolonization failed and will be unlikely in the future due to the larger distances between fragmented habitat and the lack of nature reserves with a large area of suitable habitat.

The abundance and especially the historical distribution of this species was significantly larger in the past than today. The present number of sites includes one core population with a more or less stable population and two smaller satellite populations, of which one seems to be at the brink of extinction.

Step 3 - FRP assessment

In order to achieve a long term viable metapopulation, the objective must be to restore all previously occupied areas (deteriorated as habitat by afforestation and acidification). Sustainability can be reached with two or more core populations (Weerribben, Rottige Meenthe and maybe De Wieden) of at least 1,000 individuals (based on consideration of stochasticity of annual population fluctuations; compare Table 6.1.1) and satellite populations in previously occupied areas (e.g. Bancopolder, Lindevallei, De Deelen) each with a modest population size of around 100 individuals on average, which functions altogether as one metapopulation.

Conclusions

Although estimates of population size for the Netherlands are unknown before 1950, there is evidence that the species was much more common and with a much larger distribution in the past than when the Habitats Directive came into force. Although ideal population size would be in the range of 5,000-10,000 (Ottburg & Van Swaay 2014), a feasible and viable threshold based on two current core areas and 12 (historically occupied) satellite areas, together forming one meta-population, results in:

FRP = about 1,000*2 + 12 * 100 = 3200 individuals.

Step 4 - FRR assessment

The historical range of the species was much larger with more satellite populations. We set FRR >> CV to include the FRP of a long term viable metapopulation corresponding to its former natural range in the country.

Conclusions

FRR >> CV.

References

Bink, F.A. 1986. Acid stress in Rumex hydrolapathum (Polygonaceae) and its influence on the phytophage Lycaena dispar (Lepidoptera; Lycaenidae). Oecologia 70: 447-451.

Bink, F.A. 1992. Ecologische atlas van de dagvlinders van Noordwest Europa. Schuyt & Co. Haarlem: 512 pp.

Bink, F. (2013) Dagvlinders in de Benelux. Eigen uitgave auteur.

Bos, F., M. Bosveld, D. Groenendijk, C. van Swaay, I. Wynhoff, 2006. De dagvlinders van Nederland, verspreiding en bescherming (Lepidoptera: Hesperioidea, Papilionoidea. – Nederlandse Fauna 7. Leiden. Nationaal Natuurhistorisch Museum Naturalis, KNNV Uitgeverij & European Invertebrate Survey - Nederland.

Groenendijk, D.; Swaay, C.A.M. van (2005) Profielen Vlinders en Libellen van de Habitatrichtlijn Bijlage II. Rapport VS2005.21 - Vlinderstichting, Wageningen.

Ottburg, F.G.W.A. & C.A.M. van Swaaij (2014) Gunstige referentiewaarden voor populatieomvang en verspreidingsgebied van soorten van bijlage II, IV en V van de Habitatrichtlijn. WOt-rapport 124. Wageningen, Wettelijke Onderzoekstaken Natuur & Milieu.

Swaay, C.A.M. van; Termaat, T.; Kok, J.; Huskens, K.; Poot, M. (2017) Vlinders en libellen geteld : jaarverslag 2016. Rapport VS2017.01 - De Vlinderstichting, Wageningen,

Swaay, C.A.M. van; Bink, F.A. (red.) (2000) Beschermingsplan grote vuurvlinder 2000-2004. Rapport Directie Natuurbeheer nr. 39 - Ministerie van Landbouw, Natuurbeheer en Visserij, Wageningen.

7 Vascular plants and bryophytes

7.1 Fen orchid (*Liparis loeselii*) in the Netherlands

John Janssen

Step 1.1 - Biology of the species

Growth form, phenology and propagation
Liparis loeselii is a small orchid, flowering in spring or early summer (Weeda *et al.* 1994). Germinating plants are capable to develop leaves within one year, and afterwards form a root stock in the form of a greenish bulb (giving the species its Dutch name "green bulb orchid"). The upper part of the bulb grows above ground level and therefore is capable of photosynthesis. The bulb remains over the winter and, when mature, forms a second, smaller bulb at its side. The average life span of an individual is assumed to be relatively short (up to a few years).

The flowers are small and pale yellowish green, and seem not to attract insects. Self-pollination is considered the main way of propagation, in general leading to good seed production. As a general rule, high levels of self-pollinating lead to strong genetic differentiation between populations and low variation within populations. However, a study on the Wadden Islands showed that genetically distinct populations occur (Sharudin *et al.* 2014b). A relatively high within-population variation was found. In the first place this was explained by locally high dispersal rates. Although *Liparis loeselii* has light seeds which are easily dispersed by wind (like most orchids), long-distance dispersal (over many km's) appears to be rare. Secondly, high rates of colonisations (multiple founders) from different populations leads to a diverse genetic structure. The latter is confirmed by genetic data, suggesting that new habitat (young dune slacks) is colonised from several populations (from older dune; Sharudin *et al.* 2014b).

Habitat
In Northwest-Europe *Liparis loeselii* is mainly growing in coastal dune slacks and calcareous fens. Fen and dune plants in some studies are distinguished as different varieties (var. *loeselii* and var. *ovata*, respectively), based on the length and width of leaves (Oostermeijer & Hartman 2014). In the UK and France strong genetic differentiation was found between dune and fen populations, while dune populations from France and England did not differ that much (Pillon *et al.* 2007). On the other hand, a study across Europe provided little evidence for ecotypic differentiation between populations from different habitats (Vanden Broeck *et al.* 2014). In dune slacks the species grows in early succession stages (young dune slacks), where many open spots are available and little soil development took place. In fens the species is found mainly in sites with seepage of calcareous or base-rich groundwater or in sites that are temporary flooded with calcareous or base-rich surface water.

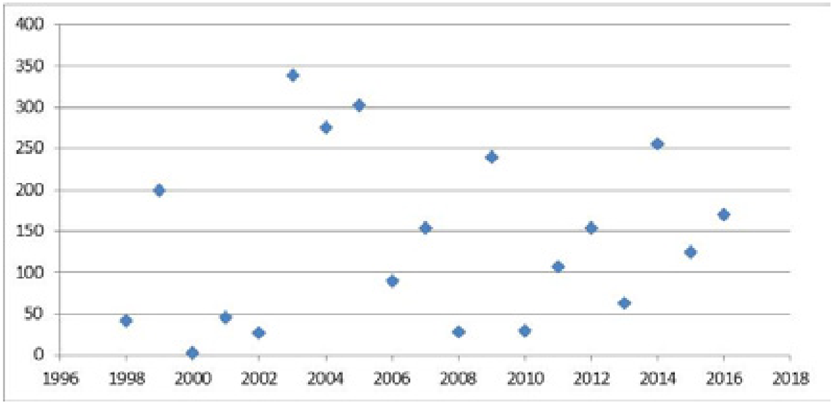


Figure 7.1.1 *Counts of flowering individuals of Liparis loeselii in Inlaag Hoofdplaat (data: Het Zeeuwse Landschap, unpublished)*

Population dynamics
Local population show strong fluctuations from year to year and certainly within a few years. This is illustrated in Figure 7.1.1 for the site Inlaag Hoofplaat in the Southwest of the Netherlands. Here, the species grows in a relatively stable habitat (a kind of calcareous dune slack). Only flowering individuals were counted, which is a usual practice for many plant species, as it requires far more effort to count non-flowering individuals, and it is impossible to count dormant individuals. In more dynamic situations the population size may vary even more, with strong increasing or decreasing numbers, due to changing of the habitat. In such dynamic landscapes the constant creation of new, suitable habitat is a requirement for maintaining a stable population (within the large fluctuations).

Step 1.2 - Spatial scale of functioning
In the Netherlands the orchid *Liparis loeselii* has two main regions of occurrence, one in the coastal dunes, and one in inland base-rich fens. The Dutch fen populations consists of two subpopulations, rather isolated from each other (Figure 2). Therefore we distinguish three meta-populations in the Netherlands: one in the coastal dunes, one in the western fen-area of Holland (provinces Zuid-Holland, Noord-Holland, and Utrecht), and one in the northern fen-area (provinces Overijssel and Friesland). A few additional, small and isolated populations exist. The coastal dune meta-population is connected to a population on the island of Borkum in Germany (Petersen 2003-2009). Possibly also exchanges occur with populations in Belgium.

In coastal dunes the species occurs mainly in rather dynamic habitats, and more rarely in low-dynamic conditions. In the dynamic dune landscape new habitat (young dune slacks) should be formed regularly to maintain a large population. In the fen area the habitats are relatively stable and low-dynamic.
Like for many other plant species, it may be difficult to count population sizes, but estimates of the number of adult plants have been carried out in several studies (Odé & Bolier 2001; Žalneravičius & Gudžinskas 2016). We try to use number of individuals for estimating favourable population sizes. An alternative may be to use relatively detailed distribution grid data as a proxy for population changes. For the assessment of the range we will use the number of occupied 5x5 km grids as a first proxy, as long-term historical data is available on this scale only. For the assessment of favourable reference range (FRR) these values will be transformed into number of 10x10 grid cells (according to Dutch co-ordinate systems).

Conclusions

We consider population category S2 appropriate for this species: species with clearly disjunct distributions and one or a few isolated (often genetically differentiated) populations at the national level.

Step 1.3 - Historical perspective: what happened to the species?

The distribution in 5x5 km^2 grid cells is given in Figure 7.1.2 for different time periods. *Liparis loeselii* was found in 110 grid cells before 1930, and in 90 afterwards, while it was found in respectively 143 and 64 before and after 1995. The main decline in range took place in the first decades of the 20th century. After 1950 only a few locations remained in the south-eastern half of the Netherlands, the sand landscape (see the second map in Figure 7.1.2). Here *Liparis loeselii* grew in brook valleys and wet heathlands (Kreutz & Dekker 2000), but in these habitats the species nowadays is extinct. Since about 1960 the species is almost exclusively known from the coastal dunes (dune slacks) and (base-rich) fen areas. The overall range has not changed significantly during the last 50 years, but nowadays slightly less grid cells within the remaining range are occupied compared to the period before 1950, especially in the western fen area.

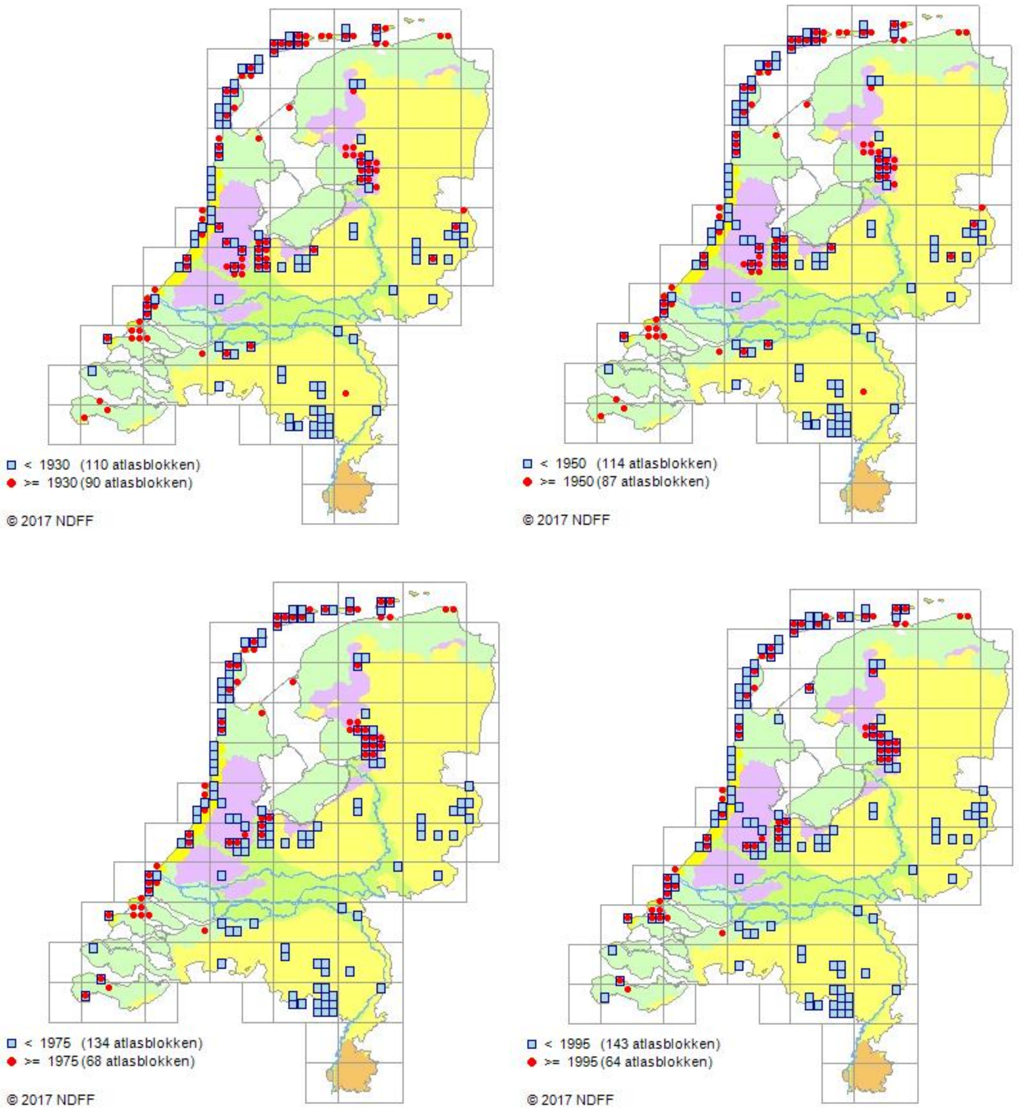


Figure 7.1.2 *Distribution of Liparis loeselii in the Netherlands in 5x5 km-grids in different periods (Floron 2017). In the last period (bottom-right) it can be seen that the present distribution (red dots) is much smaller than the historical distribution (blue grids)*

Step 1.4 - Analysis of distribution and trend

Distribution
Based on distribution maps of different periods (Figure 7.1.2) the number of occupied 5x5 km grid cells have been calculated for the period 1990-2000 (the period when the HD came into force), the current period (2005-2015) and a historical period 1950-1975, when the conditions in the fen region were still relatively better, but outside the dune and fen regions most populations had already declined. The values are given in Table 7.1.1.

In the <u>dune region</u> in the period 1990-2000 *Liparis loeselii* was recorded in 30 5x5-km-grids. This number is more-or-less stable compared to the reference period 1950-1975. Presently (period 2000-2010) the amount of grid cells is slightly higher.

In the <u>fen regions</u> in the period 1990-2000 the species was recorded in 5 grids in Holland/Utrecht and 13 grids in Friesland/Overijssel (Table 7.1.1). The distribution in Holland/Utrecht was twice as high in the reference period 1950-1975. In Friesland/Overijssel the trend in distribution grids compared to the period 1950-1975 is a small increase.

Outside these regions, the species still survived in two grid cells in the period 1990-2000, summing the total to 50 grid cells at the date when the Habitats Directive came into force.

Table 7.1.1 *Distribution in different regions and different periods, expressed in numbers of 5x5 and (between brackets) 10x10 grid cells (after data from Floron 2017)*

Distribution range in 5x5 grid cells (10x10 in brackets)				
	1950-1975	1990-2000	2005-2015	FRR
Dune region	27 (18)	30 (19)	37 (25)	30 (19)
Fen region1 (H/U)	10 (6)	5 (3)	7 (4)	8 (5)
Fen region2 (F/O)	10 (6)	13 (7)	14 (6)	13 (7)
Outside	7 (7)	2 (1)	0 (0)	2 (1)
TOTAL	*54 (37)*	*50 (30)*	*58 (35)*	*53 (32)*

Population size
Assessments of the total population size are only known since 2001. Odé & Bolier (2001) estimated the total Dutch population to be 20,000 individuals, with a maximum of 50,000 in extreme good years. A count of the overall population size in 2010 by Oostermeijer & Hartman (2014) resulted in 10,000 to 11,000 individuals, but this seems an underestimate, if other figures from the same period are considered (for example more than 10,000 individuals in that period in Lake Grevelingen according to De Kraker 2015). In several nature reserves detailed counting has been carried out since 2000, showing strong fluctuations. As an example, numbers of flowering plants in several subpopulations in Lake Grevelingen are given in Figure 7.1.3, based on data collected by De Kraker (2005-2015).

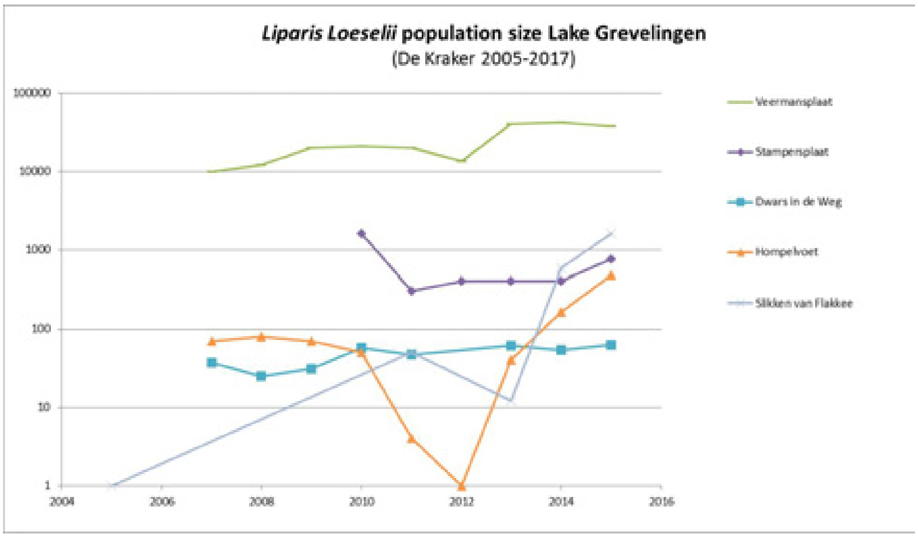


Figure 7.1.3 *Population size since 2005 on several islands and shores of Lake Grevelingen (data collected by De Kraker 2005-2015)*

Detailed data on population sizes from the dune region before 2000 is not known, but from own observations it is clear that also here large fluctuations occurred within less than a decade due to the development and decline of young succession stages in primary dune slack. An example is the most western dune slack on Schiermonnikoog. Although no exact data on population size are available, it is known that a large population of *Liparis loeselii* quickly developed around 1995, which afterwards declined due to further soil development and succession of dune slack vegetation, amongst others resulting in a more closed sward and superficial acidification of the soil. It is likely that also on the islands of Schiermonnikoog, Ameland and Terschelling the total population has decreased during the last decades, due to aging of dune slacks, even though some new populations have established themselves.

On the other large Wadden Sea islands (Texel, Vlieland) recently in young primary dune slacks large populations of *Liparis loeselii* have developed. The same happened on the Kennemerstrand (dunes near Amsterdam), where the development is the result of the construction of a large dam, perpendicular to the coastline. In front of this dam huge amounts of sand were deposited by wind and sea, resulting in the development of a broad beach on which dune slack vegetation developed. An even stronger population increase was seen in Lake Grevelingen (Figure 7.1.3), but under even more artificial conditions. The former Grevelingen estuary was dammed in 1971. The water remained saline, but the tidal regime disappeared. As a result former intertidal sand flats stood clear of water permanently, and on these new islands and shores a rapid vegetation succession occurred. This lead to the current presence of large areas with plant communities characteristic to calcareous dune slacks. Here, *Liparis loeselii* is one of the characteristic species. On one island (Veermansplaat) by far the biggest population of the Netherlands occurs: more than 30,000 flowering individuals (De Kraker 2015).

For the maintenance of the large recently established populations in the dune region mowing is the common management practice. However, on the long run the regular development of new, young succession stages of dune slacks is needed. On the Wadden Sea island of Texel and Vlieland such opportunities exist for the near future. On the Kennemerstrand new dune slacks may develop in future as well, but the situation is different in Lake Grevelingen, where the succession leads in one direction. Although succession is slowed down by grazing and mowing, and the high amount of chalk in the soil provides good prospects, for the long term a decline in *Liparis loeselii* population can be foreseen, as the conditions will ultimately become less suitable due to soil acidification, accumulation of organic material and closing of the vegetation sward. It is unknown how long the present population may be able to stabilise or even grow further.

After having made some expert assessments for data gaps, an estimate of the total population in the dune region since 2007 is given in Figure 7.1.4 (after Rusman *et al.* 2017). The population size increases, but this is mainly caused by extreme high and increasing numbers on the small island Veermansplaat in Lake Grevelingen (Figure 7.1.2). Data on population size and trends in the fen regions are less well known and are based on more estimates (Figure 7.1.5). It is likely that the population has declined much since the 1950's, but estimates from 2001 indicate a rather stable total population size, with a minimum of 6000-8000 and higher figures in a few years.

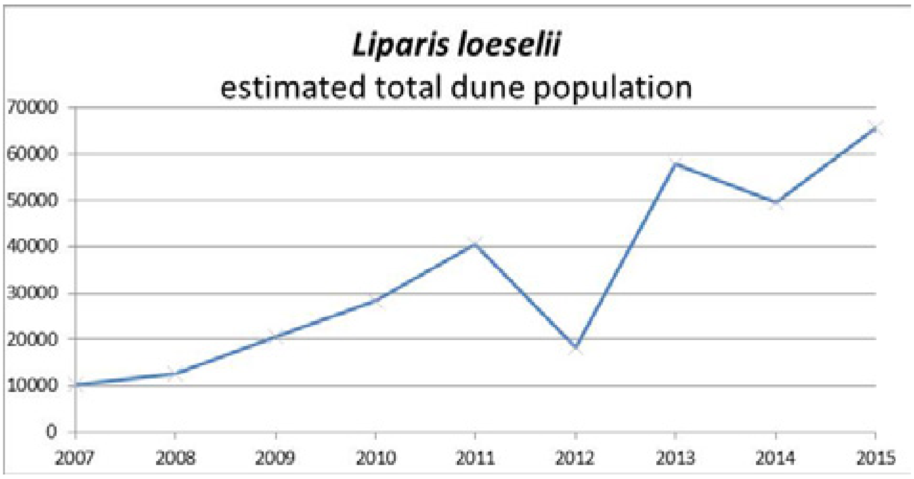


Figure 7.1.4 *Estimate of total population size in the dune region since 2006 (after Rusman et al. 2017)*

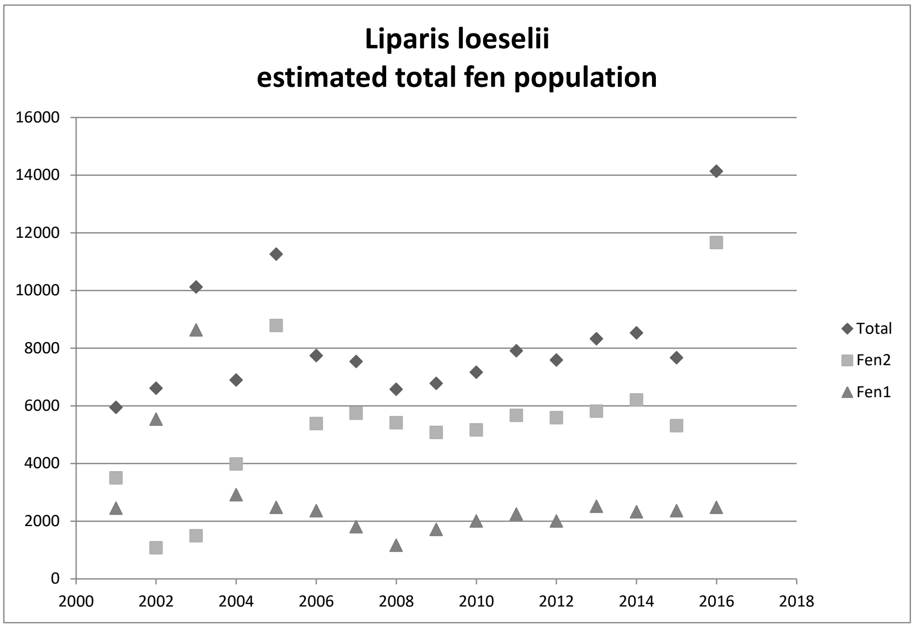


Figure 7.1.5 *Estimate of total population size in fen regions since 2001 (data based on Rusman et al. 2017)*

Conclusions

During the 20th century a strong decline in distribution of Liparis loeselii occurred. Already before 1950 the species disappeared almost completely from brook valleys and wet heathlands. The remaining distribution in coastal dunes and inland fens has not changed significantly during the last 50 years, but especially in the western fen area slightly less grid cells are occupied recently.

We conclude that FRPs should be defined separately for three more or less isolated meta-populations in the Netherlands: one in the total coastal dune region, one in the fen region of Holland/Utrecht, and one in the fen region of Friesland/Overijssel.

Outside these regions the populations of Liparis loeselii have almost completely disappeared long ago, and restorability is rather unlikely given irreversible changes in land use and abiotic conditions. However, the few populations still remaining in 1994 (when the Habitat Directive came into force), have to be included for the assessment of FRP and FRR.

The distribution is stable since about 1950, except for the fen region of Holland and Utrecht, in which a small decline occurred. Data on population sizes is not available for a long period but it is likely that large declines occurred in the fen regions, while populations fluctuated a lot and possibly reached relatively high numbers currently, in the coastal dunes.

Step 2.1 - FRP assessment

No detailed total population size is known from the period the Habitats Directive came into force. It is however likely that it was similar or lower than the first overall estimate by Odé & Bollier (2001) of 20,000-50,000 individuals. Therefore we use this range in 2001 as a reference value (FRP)[3]. However, additional, we require a minimum population in all of the three regions with different meta-populations, as is given in the conclusions below.

Conclusions

The FRP is the sum of the values in 2001 for the different regions, and is set at 20,000 (to 50,000) individuals.

For the different regions the following minimum values of the population size (partial FRPs) are required:

Dune region	14000
Fen region1 (Ut/Ho)	2000
Fen region2 (Fr/Ov)	5000
Other regions	presence

Step 2.2 - FRR assessment

We consider the distribution and range in the coastal dune region and the fen region of Friesland/Overijssel at the date the HD came into force sufficient to maintain its reference population size. However in the fen region of Utrecht/Holland an increase in distribution grid cells is needed to guarantee long-term survival of the (relatively small) populations. Restoration seems possible as some new grid cells were recorded in the period 2005-2015. We roughly estimate that restoration of at least half of the decline since 1950-1975 is necessary to reach stable population trends. Whether such a restoration is enough has to be monitored. The FRR for the Utrecht/Holland population therefore is 7 grid cells of 10x10 km^2.

The indicated total FRR is 32 grid cells of 10x10 km^2, which is slightly higher than the FRR of 25 distribution grid cells assessed by Odé in Ottburg & Van Swaay (2014). It should be noted however that the provided distribution map in this latter study is incomplete.

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[3] Note that the proposed FRP by Odé (in Ottburg & van Swaay 2014) was 20,000 individuals.

Conclusions

FRR =3200 km^2 (32 grid cells of 10x10 km^2).

with the following minimum values (partial FRRs) for the different regions:

Dune region	19 grid cells
Fen region1 (Ut/Ho)	5 grid cells
Fen region2 (Fr/Ov)	7 grid cells
Other regions	1 grid cells

References

Bolier, A. & B. Odé (2001). Groenknolorchis op de kaart. Gorteria 29: 33-37.

De Kraker, C. (2005). Kartering meetsoorten Grevelingen 2005. Ecologisch adviesbureau Sandvicensis, Burgh-Haamstede.

De Kraker, C. (2009). Grevelingenverslag 2009. Ecologisch adviesbureau Sandvicensis, Burgh-Haamstede.

De Kraker, C. (2012). Vegetatie van de Grevelingen - kartering meetsoorten 2009-2011. Ecologisch adviesbureau Sandvicensis, Burgh-Haamstede.

De Kraker, C. (2015). Grevelingenverslag 2015. Ecologisch adviesbureau Sandvicensis, Burgh-Haamstede.

Floron (2017). https://www.verspreidingsatlas.nl/vaatplanten; date 16-01-2017

Kreutz, C. A. J. & H. Dekker (2000). De orchideeën van Nederland: ecologie, verspreiding, bedreiging, beheer. Seckel, Raalte, 512 pp.

Oostermeijer, G. & Y. Hartman (2014). Metapopulation dynamics of Liparis loeselii. Acta Oecologica 60: 30-39.

Petersen, J. (2003, 2006, 2009). Monitoring der Vorkommen von Liparis loeselii auf Borkum im Rahmen der FFH-Berichtspflicht. Unveröff. Gutachten im Auftrag des NLWKN.

Ottburg, F.G.W.A. & C.A.M. van Swaay (2014, red.). Gunstige referentiewaarden voor populatieomvang en verspreidingsgebied van soorten van bijlage II, IV en V van de Habitatrichtlijn. WOt-technical report 124, Wettelijke Onderzoekstaken Natuur & Milieu, Wageningen, pp. 253-255.

Pillon, Y., F. Qamaruz-Zaman, M.F. Fay, F. Hendoux & Y. Piquot (2007). Genetic diversity and ecological differentiation in the endangered fen orchid (Liparis loeselii). Conservation Genetics 8: 177–184.

Rusman, Q., J.A.M. Janssen & A. Corporaal (2017). Een orchideetje meer of minder. In: Schaminée, J.H.J. & J.A.M. Janssen (eds). Buigen of barsten. Beschouwingen over vegetatiedynamiek. Vegetatiekundige monografieën 7. In prep.

Shahrudin, R., A. van de Craats, A.P. Grootjans & A. Kooijman (2014a). Window of opportunity for Liparis loeselii loeselii populations during coastal dune-slack succession. In: Sharudin, R., Do we really need management to preserve pioneer stages in wet dune slacks? PhD Thesis, Rijksuniversiteit Groningen, pp. 47-58.

Shahrudin, R., S. van der Meer, P. Meirmans, A.P. Grootjans & G. Oostermeijer (2014b). Genetic structure and dispersal events in a metapopulation of self-pollinating orchid in dynamic coastal wetlands. In: Sharudin, R., Do we really need management to preserve pioneer stages in wet dune slacks? PhD Thesis, Rijksuniversiteit Groningen, pp. 59-70.

Vanden Broeck, A., W. Van Landuyt, K. Cox, L. De Bruyn, R. Gyselings, G. Oostermeijer, B. Valentin, G. Bozic, B. Dolinar, Z. Illyés & J. Mergeay (2014). High levels of effective long-distance dispersal may blur ecotypic divergence in a rare terrestrial orchid. BMC Ecology 14(20): 1-14.

Žalneravičius, E. & Z. Gudžinskas (2016) Assessment of the data on distribution, habitats and population size of Liparis loeselii (Orchidaceae) in Lithuania. Botanica Lithuanica 22(1): 3–15.

7.2 Varnished-Hook moss (*Hamatocaulis vernicosus*) in the Netherlands

Rienk-Jan Bijlsma

Step 1.1 - Biology of the species

Hamatocaulis vernicosus (=Drepanocladus vernicosus) is a rather large pleurocarpous bryophyte with hooked shoot tips (hence its English name) related to and often growing together with other so-called ‘fen mosses’. The inclusion of this rather elusive species on Annex II of the Habitats Directive generated a lot of research to clarify its distribution and ecological requirements.

A molecular study revealed two cryptic species within *Hamatocaulis vernicosus*, a widespread species in the boreal zone and a more southern species (Hedenäs & Eldenäs, 2007). As discussed by these authors, the study is significant from the conservation point of view because wetlands in the boreal zone are still in a relatively natural state, while those in southern Sweden, and even more the wetlands further south in Europe, are much more strongly negatively affected or destroyed. I assume that the Dutch populatons belong to southern ‘species’.

Hamatocaulis vernicosus belongs to a group of ‘rich fen’ species found in wetlands with relatively high pH, EC, Ca, Mg and HCO_3. Species of *Tomentypnum*, *Scorpidium* and *Calliergon* belong to this group as well (Hedenäs & Kooijman, 1996). Within this group the habitat of *Hamatocaulis* is uniquely characterized by high Fe-concentrations in soil pore waters and mostly N-limited conditions with low Ca:Fe ratios and labile N concentrations, and high concentrations of plant available P and Fe-bound P (Stechová *et al.*, 2012; Pawlikowski *et al.*, 2013; Cusell *et al.*, 2014). These requirements restrict the ecological feasibility of habitat restoration to specific (N-limited) biogeochemical conditions.

Long-distance dispersal is probably limited by the fact that sporophytes of the dioicous *Hamatocaulis* are absent or very rare in many parts of its range and e.g. have never been found in the Netherlands (Touw & Rubers, 1989) and France (Pépin *et al.*, 2013). Only in Scandinavia fruiting in this species is considered ‘just’ rare (Hedenäs & Hallingbäck, 2014). Within sites, the spatial dynamics of *Hamatocaulis* mainly depends on vegetative reproduction by clonal growth and dispersal by stem fragments, e.g. due to mowing or flooding.

Step 1.2 - Spatial scale of functioning

As a result of its specific habitat requirements and clonal growth, *Hamatocaulis* generally has a scattered distribution and population clusters within sites.

Conclusions

Hamatocaulis is an uncommon plant species with scattered (often fragmented) distribution (population category S6).

Step 1.3 - Historical perspective: what happened to the species?

The fate of *Hamatocaulis* in the Netherlands is exemplary for its history in the European lowland where the species is very rare nowadays (Figure 7.2.1). It occurs in neutral flushes and fens and particularly likes domed springheads and seepage areas, where alkaline water breaks through acidic peat, or areas where alkaline flushes spread onto acidic ground (Atherton *et al.*, 2010). In the Netherlands most fens in general and rich fens in particular have been reclaimed and cultivated well before the 1950s (Kooijman, 1992). The remaining sites face anthropogenic acidification and eutrophication by high levels of N-deposition (Cusell *et al.*, 2014). Based on herbarium records, 40% of the known distribution was already lost before 1900 (Figure 7.2.2). Despite the very low recording intensity of bryophytes in the 19th century, *Hamatocaulis* was found throughout the region with higher sandy soils and must have been much more common than documented in that period. After 1965 the species was considered extinct in the Netherlands until 1996 when a large new locality was found (Van Tweel & Van Wirdum, 1999).

Figure 7.2.1 *Distribution of Hamatocaulis vernicosus in north-western Europe (source: http://art17.eionet.europa.eu/article17/reports2012/species/summary/?period=3&group=Non-vascular+plants&subject=Drepanocladus+vernicosus&region=)*

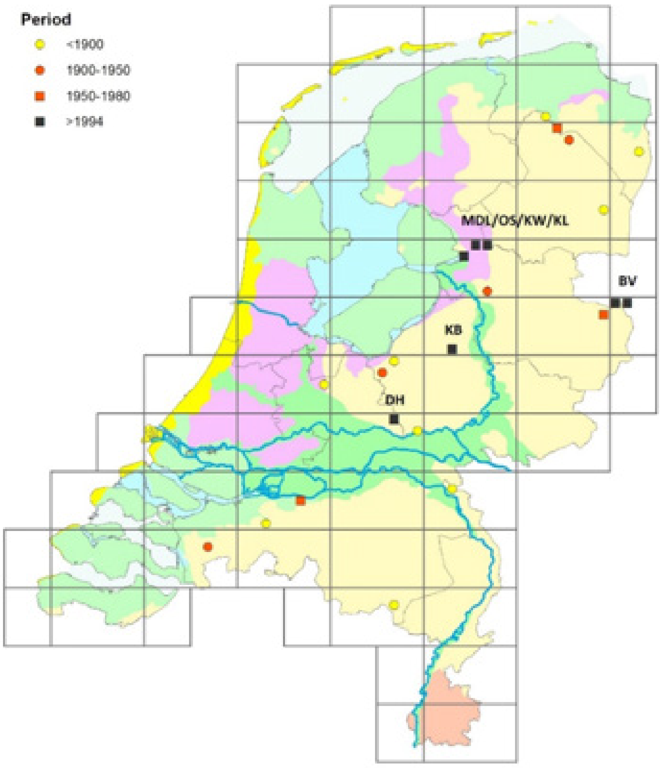

Figure 7.2.2 *Distribution of Hamatocaulis in the Netherlands (5x5 km squares, based on herbarium collections). For location labels, see 1.4 (based on https://www.verspreidingsatlas.nl/2710#)*

Step 1.4 - Analysis of distribution and trends

The northern and southern parts of the historical range were lost after 1952 and 1965, respectively. Until 2004 the distribution was restricted to just one site (Table 1, MDL). Between 2004 and 2016 six new, smaller sites have been discovered, three in the same region (OS, KW and KL; Figure 2), and three isolated sites in the eastern and central part of the country. The small BV-site was discovered in

2013 but appeared to be lost in 2016 already. In 2014 the new KB locality was discovered in a restoration project area in a former agricultural landscape dating from 1997-98; this can only be explained by long-distance dispersal and demonstrates that succesful restoration is possible despite many apparently limiting factors.

Since 2004 *Hamatocaulis* is monitored in all known sites within and outside Natura 2000 (Table 1). Only the largest site (MDL) shows a positive trend while all other sites are much smaller and show more or less stable or negative trends. Some of the smaller sites (e.g. KW and DH) are probably not recently established but overlooked during surveys in the past and suggest that viable ‘populations’ occupying at least 20-25 10x10 m squares can withstand short-term environmental fluctuations.

Table 7.2.1 *Number of 10x10 m squares (‘population size’) occupied by Hamatocaulis per site (compare Figure 7.2.2). MDL Mepperler Dieplanden; KW Kiersche Wieden; KL Kikkerland; OS: Oude Stroom; DH: De Hel; BV: Brecklenkampse Veld; KB Korte Broek (from Van Tweel & Sparrius, 2016)*

Year	MDL	OS	KW	KL	DH	BV	KB
2004	388						
2007	643	15	16				
2010	797	30	21		23		
2013	1097	24	15	4	20	8	
2016	1354	10	12	4	12	0	17

Conclusions

Most of the distribution and habitat of *Hamatocaulis* in the Netherlands was lost before 1950 or perhaps already before 1900. The northern and southern parts of the historical range were lost after 1952 and 1965, respectively. Up to 2004 only one site with a large and viable population was known. Afterwards, six much smaller populations were discovered, three of which in the same region as the first population. Data from short-term monitoring of the smaller sites suggest that a ‘viable population’ must occupy at least 20-25 10x10 m squares to cope with short-term environmental fluctuations.

Step 2.1 - FRP assessment

The strong decline in the number of sites and the loss of large parts of the historical range asks for a simultaneous assessment of FRVs such that the FRP allows for the restoration of (a part of) the range and the FRR is large enough to include the required FRP.

Practically nothing is known about the dynamics, survival and viability of *Hamatocaulis* ‘populations’ (mostly clones) in relation to environmental fluctuations and succession, even in the case of adequate management preventing the encroachment of shrubs and woodland. However, the available sparse data show that 1) *Hamatocaulis* is able to double its occupancy within a decade by fragmentation and clonal growth under suitable conditions (site MDL), 2) *Hamatocaulis* is still able to establish succesfully by long-distance dispersal (site KB), 3) Relatively small populations (20-25 10x10 m squares) can withstand short-term environmental fluctuations (sites OS, KW, DH) and 4) Restoration of *Hamatocaulis* habitat is feasible in wetland situations with relatively high concentrations of available P and Fe-bound P as long as N is limiting (site KB and see step 1.1).

In order to include spatial risk spreading such that extreme local events don’t result in the sudden extinction of a population, I assume a safety factor of 5 which gives a minimum ‘viable population’ size of 1-1,5 ha. This means that only the MDL-site (13-14 ha; Table 1) is clearly safe and at least some of the other populations must be enlarged or connected to (new) adjacent populations. Moreover, to restore a significant part of the historical range additional populations are necessary which seems feasible only in the northern part of the country with a relatively low N-deposition and opportunities in former ‘chains’ of historical locations (see Figure 7.2.2). This requires (at least) two populations at all existing sites plus two locations in the northern part of the country (provinces of Drente, Groningen, Friesland), giving a total of 10 populations, each 1-1,5 ha[4].

Conclusions

FRP = 10 populations, each occupying 1-1,5 ha = 100,000- 150,000 m^2.

Step 2.2 - FRR assessment

The required 10 populations have been assigned to at least five locations, including the four existing ones, resulting in at least 5 10x10 km squares. The derivation of the FRR requires that in assessing the range relative to this FRR, the spatial configuration must be taken into account as well.

Conclusions

FRR = at least 5 locations to encompass 10 populations = 500 km^2.

References

Cusell, C., A.M. Kooijman & L.P.M. Lamers (2014) Nitrogen or phosphorus limitation in rich fens? - Edaphic differences explain contrasting results in vegetation development after fertilization. Plant Soil 384: 153–168.

Hedenäs, L.& A.M. Kooijman (1996) Phylogeny and habitat adaptations within a monophyletic group of wetland moss genera (Amblystegiaceae). Pl.Syst.Evol. 199: 33–52.

Hedenäs, L. & P. Eldenäs (2007) Cryptic speciation, habitat differentiation, and geography in Hamatocaulis vernicosus (Calliergonaceae, Bryophyta). Pl.Syst.Evol. 268: 131–145.

Pawlikowski, P., K. Abramczyk, A. Szczepaniuk & Ł. Kozub (2013) Nitrogen:phosphorus ratio as the main ecological determinant of the differences in the species composition of brown-moss rich fens in north-eastern Poland. Preslia 85: 349–367.

Pépin, F,. V. Hugonnot & J. Celle (2013) Sex ratio patterns and fertility of Hamatocaulis vernicosus (Mitt.) Hedenäs at different spatial scales. J.Bryol. 35(1): 20-26.

Sparrius, L.B. (2014) Geel schorpioenmos (Hamatocaulis vernicosus). In F. G.W.A. Ottburg & C.A.M. van Swaay (red.), Gunstige referentiewaarden voor populatieomvang en verspreidingsgebied van soorten van bijlage II, IV en V van de Habitatrichtlijn. Wageningen, Wettelijke Onderzoekstaken Natuur & Milieu, WOt-rapport 124; 250-252.

Štechová, T., J. Kučera & P. Šmilauer (2012) Factors affecting population size and vitality of Hamatocaulis vernicosus (Mitt.) Hedenäs (Calliergonaceae, Musci). Wetl.Ecol.Manage. 20: 329–339.

Tweel, M.J. van & G. van Wirdum (1999) Scorpidium vernicosum in de Meppelerdieplanden. Buxbaumiella 48: 21-23.

Tweel, M.J. van & L.B. Sparrius (2016) NEM Meetnet Geel schorpioenmos. Rapportage meetronde 2016. BLWG Rapport 2016.02. Bryologische en Lichenologische Werkgroep, Breda.

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[4] The current FRP, purely based on expert opinion, is based on 10 populations with a total occupied area of 150,000 m^2 (Sparrius, 2014).

8 Marine habitats

Susan Gubbay

8.1 Sandbanks which are slightly covered by sea water all the time (1110) in the United Kingdom

Step 1.1 - Ecology of the habitat
Marine banks are defined as elevations of the seafloor of relatively large areas, surrounded by deeper water (American Geological Institute 1974). The term is generally used to refer to sedimentary accumulations and, in the case of sandbanks, they are features that consist mainly of sand size sediments (grain sizes of between 0.06-2 mm).

Sandbanks are formed and maintained by different types of marine flows, where there is abundant sand and where currents are strong enough to move sediment. Some may owe their morphology to the topography of underlying hard substrate (Dyer & Huntley 1999; Velegrakis *et al.* 2001). A descriptive classification scheme developed by Dyer & Huntley (1999) distinguishes between linear, irregular and shore-attached banks in different geographical situations. This has been developed further by Kenyon & Cooper (2005) into a classification based on patterns of sediment transport (Figure 8.1.1).

Sandbanks occur in coastal and shelf seas as well as in deeper areas of the continental margin although the Habitats Directive Annex 1 type is limited to shallow sandbanks, typically less than 20m depth. The definition in the Interpretation Manual of European Union Habitats (European Commission 2013) is as follows:

> *Sandbanks are elevated, elongated, rounded or irregular topographic features, permanently submerged and predominantly surrounded by deeper water. They consist mainly of sandy sediments, but larger grain sizes, including boulders and cobbles, or smaller grain sizes including mud may also be present on a sandbank. Banks where sandy sediments occur in a layer over hard substrata are classed as sandbanks if the associated biota are dependent on the sand rather than on the underlying hard substrata.*
> *"Slightly covered by sea water all the time" means that above a sandbank the water depth is seldom more than 20 m below chart datum. Sandbanks can, however, extend beneath 20 m below chart datum. It can, therefore, be appropriate to include in designations such areas where they are part of the feature and host its biological assemblages.*

The diversity and types of community associated with this habitat are determined particularly by sediment type together with a variety of other physical, chemical and hydrographic factors. These include geographical location (influencing water temperature), the relative exposure of the coast (from wave-exposed open coasts to tide-swept coasts or sheltered inlets and estuaries), the topographical structure of the habitat, and differences in the depth, turbidity and salinity of the surrounding water (JNCC 2017). Where sandbanks are extensions of shallow coastal sediments adjacent to the coastline they may support similar soft sediment communities (Kaiser *et al.* 2004).

Communities found on sandbank crests are predominantly those typical of mobile sediment environments and tend to have low diversity. Fauna such as polychaete (cat worms) and amphipod (shrimp-like crustaceans) thrive in this environment as they are able to rapidly re-bury themselves. Animals like hermit crabs, flatfish and starfish also live on top of the sandbank (JNCC 2017). In a study of sandbanks in the Irish Sea off the coastline of Wales (Kaiser *et al.* 2004), the distinct sandbanks were typified by low species diversity and shared indicator species such as the weever fish *Echichthys vipera*, greater and lesser sandeels, the goby *Pomatoschistus minutus* the shrimp *Philocheras trispinosus*, the starfish *Asterias rubens*, and the hermit crab *Pagurus bernhardus*.

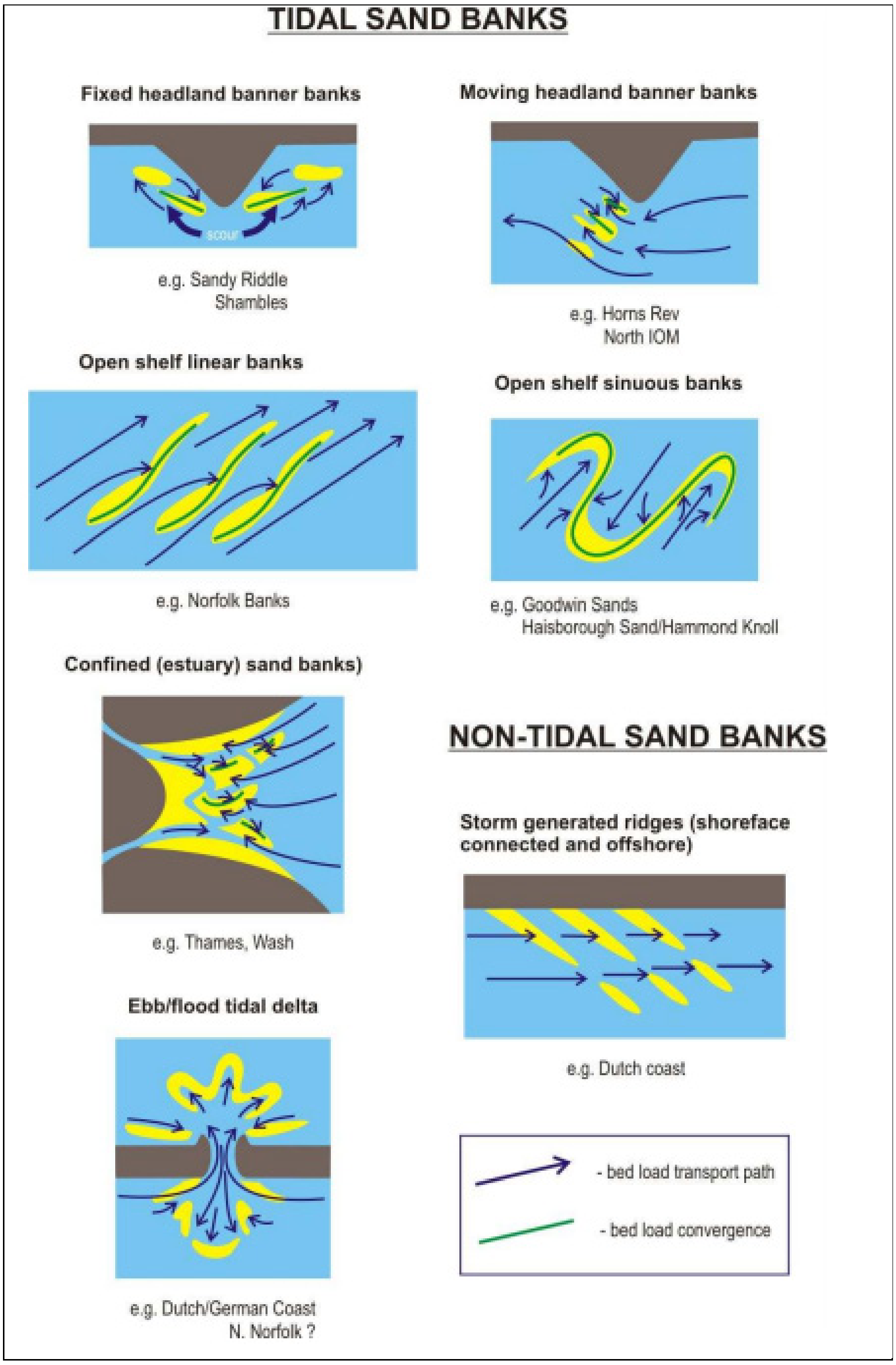

Figure 8.1.1 *Classification of sand banks (modified from Dyer & Huntley 1999) with their relation to the local and regional sand transport paths (from Kenyon & Cooper 2005)*

Troughs or areas between banks generally contain more stable gravelly sediments and support diverse infaunal and epifaunal communities. Here sediment movement is reduced and therefore the areas support an abundance of attached bryozoans, hydroids and sea anemones. In the Southern North Sea, there have been frequent observations of biogenic reefs created by the Ross worm *Sabellaria spinulosa* in association with sandbanks (JNCC 2017).

Where conditions are suitable, sandbanks also support plant communities. In the Mediterranean, these include communities characterised by the marine Angiosperm *Cymodocea nodosa* together with photophilic species of algae living on the leaves, and usually associated with *Posidonia* beds. In the Baltic these include communities of *Zostera* species, *Potamogeton* species, *Ruppia* species and *Charophytes*. In the North Sea *Zostera* species and free living species of the *Corallinacea* family are found on these sandbanks, and around the Macaronesian islands these communities consist of *C. nodosa* and *Zostera noltei* (European Commission 2013). However, on most sandbank no macrophytes are found.

Shallow sandbanks are used as resting and foraging areas by seabirds feeding on small pelagic fish and macrobenthos that are associated with the sandbanks or around the frontal systems which develop in their vicinity (e.g. Skov *et al.* 1995; Camphuysen *et al.* 2011; Boedeker *et al.* 2006; Skov *et al.* 2016). These features can also attract foraging seals (e.g. Thompson & Miller 1990; Jones *et al.* 2013). Sandbanks can play an important role in the protection of the coast by dissipating incoming wave energy and consequently acting as natural coastal defenses. They can also supply or store sediments.

Hydrographic charts are an important source of information on the location and distribution of sublittoral sandbanks. These were used together with information in the scientific literature to compile a European Sandbanks Data Base which identified 361 submerged sandbanks that potentially fit the description under the Habitats Directive (Jones 2001). The majority (>50%) were in UK waters, followed by France and Spain. In response to obligations under the Habitats Directive, Member States have undertaken dedicated surveys to locate, map, sample, and monitor Annex 1 marine habitats, including sandbanks. When combined with other sources of information, such as environmental impact assessments for offshore industries, such as marine renewables and marine aggregate extraction, this provides more detailed information about this habitat type. Two examples are given in Figure 8.1.2 and 8.1.3.

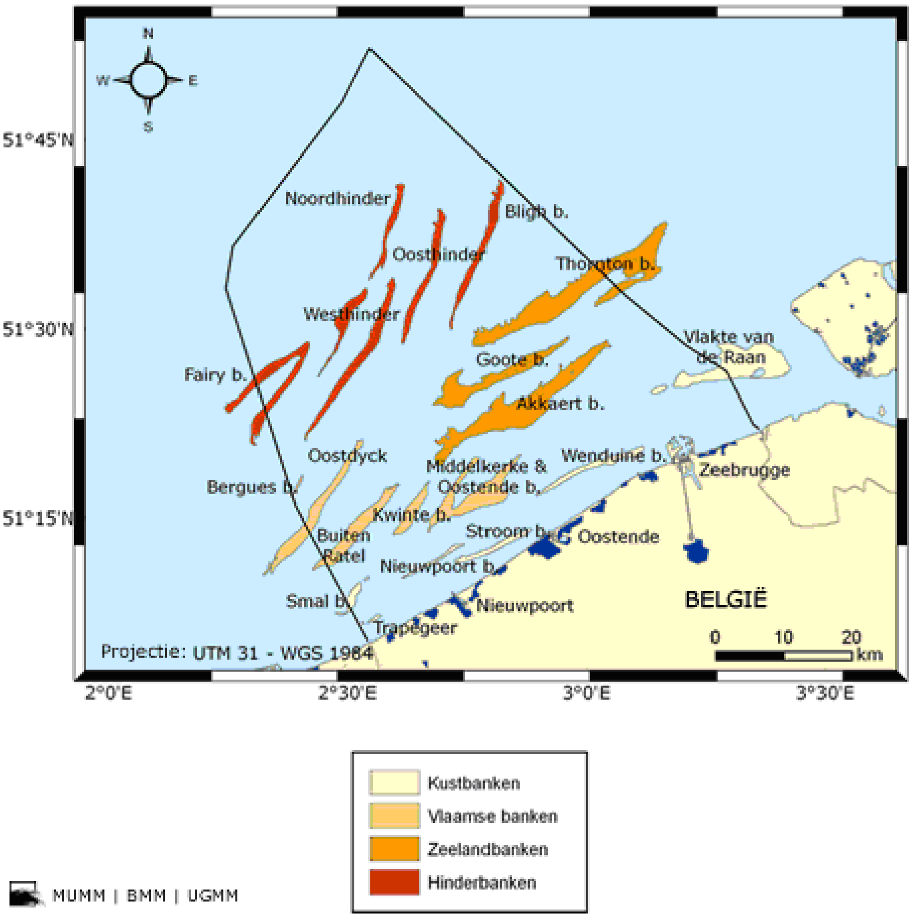

Figure 8.1.2 *Major Sandbank features in the Belgium EEZ (from Klein 2006)*

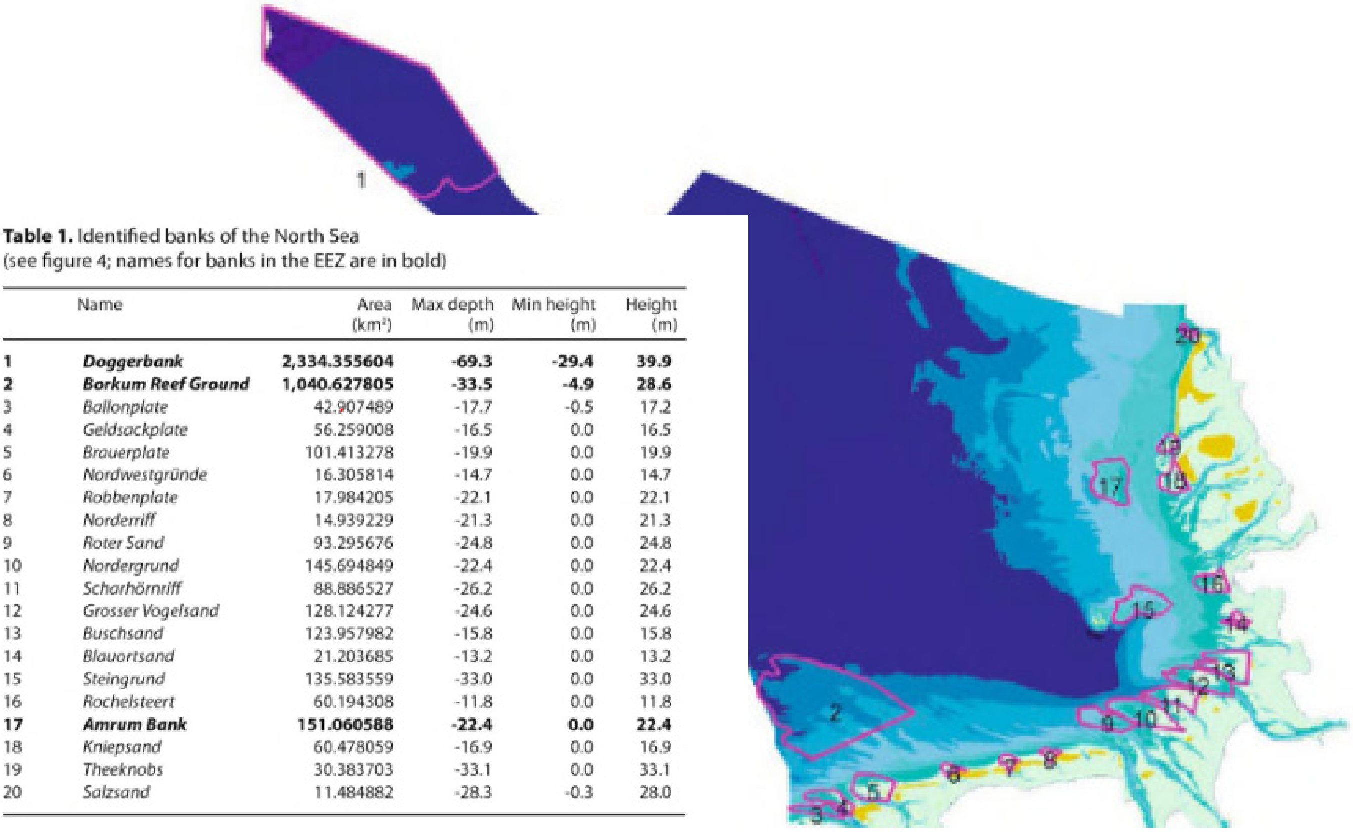

Table 1. Identified banks of the North Sea
(see figure 4; names for banks in the EEZ are in bold)

	Name	Area (km²)	Max depth (m)	Min height (m)	Height (m)
1	***Doggerbank***	**2,334.355604**	**-69.3**	**-29.4**	**39.9**
2	***Borkum Reef Ground***	**1,040.627805**	**-33.5**	**-4.9**	**28.6**
3	*Ballonplate*	42.907489	-17.7	-0.5	17.2
4	*Geldsackplate*	56.259008	-16.5	0.0	16.5
5	*Brauerplate*	101.413278	-19.9	0.0	19.9
6	*Nordwestgründe*	16.305814	-14.7	0.0	14.7
7	*Robbenplate*	17.984205	-22.1	0.0	22.1
8	*Norderriff*	14.939229	-21.3	0.0	21.3
9	*Roter Sand*	93.295676	-24.8	0.0	24.8
10	*Nordergrund*	145.694849	-22.4	0.0	22.4
11	*Scharhörnriff*	88.886527	-26.2	0.0	26.2
12	*Grosser Vogelsand*	128.124277	-24.6	0.0	24.6
13	*Buschsand*	123.957982	-15.8	0.0	15.8
14	*Blauortsand*	21.203685	-13.2	0.0	13.2
15	*Steingrund*	135.583559	-33.0	0.0	33.0
16	*Rochelsteert*	60.194308	-11.8	0.0	11.8
17	***Amrum Bank***	**151.060588**	**-22.4**	**0.0**	**22.4**
18	*Kniepsand*	60.478059	-16.9	0.0	16.9
19	*Theeknobs*	30.383703	-33.1	0.0	33.1
20	*Salzsand*	11.484882	-28.3	-0.3	28.0

Figure 8.1.3 *Submarine banks in the German North Sea (from Klein 2006)*

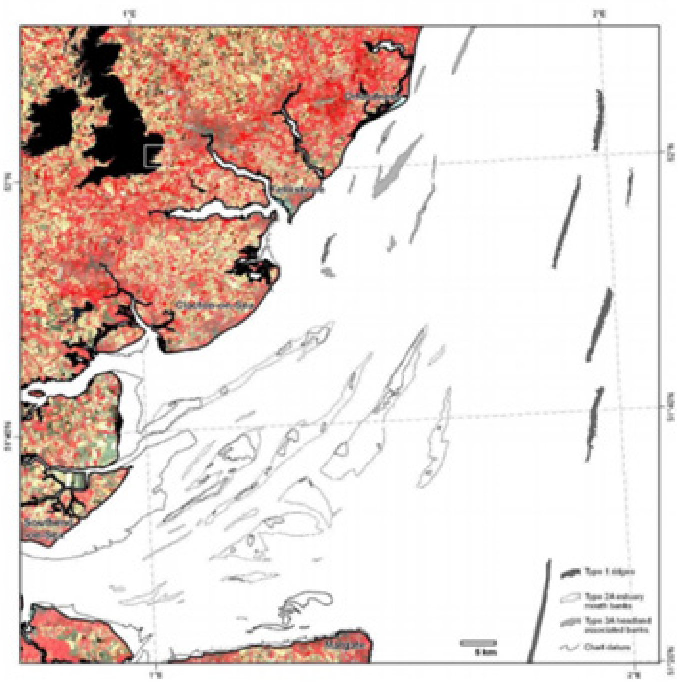

Figure 8.1.4 *The main bank features in the Outer Thames estuary (from Burningham & French 2008, with typology based on Dyer & Huntley 1999)*

Step 1.2 - Spatial scale of functioning

The development, maintenance and erosion or loss of sandbanks takes place at a variety of spatial scales. These processes may also be interconnected with other habitat features and therefore function at an ecosystem scale. For example, sandbanks can be found in association with several other Annex 1 marine habitats: Mudflats and sandflats not covered by seawater at low tide (1140), *Posidonia* beds (1120) and Reefs (1170). Sandbanks may also be a component part of Estuaries (1130) and Large shallow inlets and bays (1160).

Tidal sandbanks are the largest of the offshore rhythmic sedimentary features. They can be many kilometers in length, a spacing of up to 10km and a height of several tens of meters (Dyer & Huntley 1999). Broken Bank, one of the outermost of the Norfolk Banks in the southern North Sea, is an example of a large sandbank. This bank lies 67 km offshore and has dimensions typical of the tidally controlled open-shelf linear sandbanks of the outer Norfolk Banks area at about 30 km long, 1 km wide and with relief around 30 m above the surrounding seafloor (Schratzberger & Larcombe 2014). Groups of sandbanks can cover many square kilometers. Those found off the coast of Belgium (Figure 2) and in the outer Thames estuary (Figure 8.1.4) are well known examples.

Conclusions

Shallow sublittoral sandbanks are formed and maintained by different types of marine flows, where there is abundant sand, and where currents are strong enough to move sediment. They support a variety of benthic communities and may exhibit successional stages depending on the developmental stage of the physical habitat. They show considerable variation in size and volume. Because of the spatial scale of key processes influencing shallow sandbanks, they can be categorized as macro-habitats. Further distinction is also possible. Close to the coast, where shallow sublittoral sandbanks are part of a habitat complex, such as an estuary, the quality of the sandbank feature can be dependent on its function as part of the broader ecosystem. Under such circumstances, they may be best classified as a component of a macro-habitat (1b), whereas large offshore sandbanks or groups of sandbanks are more likely to be a macro-habitat (1a).

A reference-based method, using known area and/or distribution is considered the most appropriate for setting FRVs. The geographical range of sandbank features and salinity gradient in the Baltic Sea means there are differences in the associated communities but no obvious patterns in the distribution of sandbank type with latitude. It is therefore recommended that assessment is carried out at a national level.

Step 1.3 - Historical perspective: what happened to the habitat?

A historical perspective of changes in the shallow sandbank habitats around the United Kingdom needs to take account of the fact that they can be very dynamic features, change substantially in response to natural processes, and evolve over both short and long time scales. Any anthropogenic impacts therefore need to be viewed within the context of natural variability.

Sediment supply as well as tide, wind and wave action, and the interactions between them, determine the development of sandbanks. At one end of the spectrum a sandbank may be relict, formed under different hydraulic conditions than those existing presently and therefore not being affected actively in modern sedimentary process. The physical character of these sandbanks is unlikely to change significantly, even over decades. Others may be active where there is dynamic interaction between banks and the present hydrodynamic regime, like the shallow sandbanks in the southern North Sea (Jones 2001).

Sandbanks can be mobile features, the rate depending on the type of bank and the stage of its development (active or moribund). In the greater Thames estuary, historical changes of seabed features over 400 years derived from analysis of published bathymetric charts reveal both spatial and temporal variability. There is evidence of progressive long-term accretion and degradation as well as stepped or cyclic variability. There have been shifts in bank and channel configuration, particularly in the case of sandbanks at the estuary mouth (Burningham & French 2008). The high tidal currents in many wide estuaries mean that banks are particularly dynamic in such situations. This is illustrated by changes to bank morphology in the inner Thames where a 3km long sand bank has developed in about 30 years (Cloet 1972 in Kenyon and Cooper, 2005).

Sandbanks can also go through a cycle from an active to moribund state, for example as sea level rises and they are left stranded in weak currents (Kenyon & Cooper 2005). It is also the case that although the main feature may not change over human time there can be regular changes in the topography of the crest and superimposed bedforms (Vanaverbeke *et al.* 2000).

Step 1.4 - Analysis of distribution and trends

Repeat surveys have been carried out on some sandbanks and any long-term changes in associated species documented. On the Dogger Bank for example, where some of the earliest surveys were carried out in the early 1950's there was a considerable increase in the number of opportunistic short-living, depositing feeding polychaetes between the 1950's and the 1980's. By contrast the individual numbers of long-living bivalves such as *Ensis ensis* and *Spisula subtruncata* decreased. There was an overall decline in biodiversity although the biomass had increased over most of the bank (up to 8 times higher in places) when compared to 1950-54, except in the north-eastern part where there was a 30% reduction (Kröncke 1992). Subsequent survey work in 2000 showed that large-scale spatial distribution of the macrofauna communities had hardly changed between 1986-2000 but an increase in abundance and regional changes in distribution of various species with a southern distribution (Kröncke *et al.* 2011).

Likely factors discussed in relation to the earlier changes were seabed disturbance by fishing gears, increasing eutrophication and high concentrations of lead and cadmium. The more recent changes appear most likely related to higher sea temperatures with the increase in abundance of ophiurid *A.brachiata* on the Dogger Bank reported in 2000 a possible response to higher water temperatures since the 1980s and higher sediment mobility.

There have also been studies to investigate predicted and actual changes in response to human activities as large scale interventions such as sand extraction, can interfere with the complex hydrodynamic and morphodynamic processes (Brière *et al.* 2010). Nash Bank and Helwick Bank in the Bristol Channel have been subject to commercial dredging for aggregates and in both cases the volume of sediment in the sandbank has decreased. At Nash Bank dredging is estimated to have been responsible for around one third of the reduction over a 10-year period. In contrast, at Helwick Bank, the volume of the bank has decreased over 8 years, but at a rate six times greater than the loss due to dredging and therefore natural processes dominate here (Lewis *et al.* 2015).

Conclusions

Sandbanks can be highly dynamic features that change in character, move, erode or expand in response to natural processes such as the action of storms, tides and wave action. They may also be relict features where the main physical characteristics remain unchanged over decades. There are some historical sources of information about the location, size and movement of sandbanks in UK waters, particularly in situations where there are implications for navigation or where the features are relevant to fishing activity. There has been more detailed mapping, sampling and monitoring of sandbank features and their associated communities in recent decades but this is far from comprehensive. The main human activities that have been identified as potentially having impacts on sandbanks are eutrophication, aggregate extraction, dredging, the impact of bottom fishing gears and construction projects such as offshore windfarms that may have a direct impact on the features themselves or alter the hydrodynamic conditions that influence the status of sandbanks.

Changes in both the physical and biological characteristics of some sandbanks have been described and variously attributed to natural processes, human activity, or a combination of both. Whilst there have been localized effects there is no clear trend in the physical characteristics, but evidence that there have been some changes in the associated benthic communities in particular localities. The natural range of this habitat appears to be intact.

Step 2.1 - FRA assessment

The potential locations of shallow sublittoral sandbank habitat in UK waters have been identified using depth and slope information in combination with sediment data (Figure 8.1.5)[5]. This includes multibeam survey data of specific sandbank features, EUSeaMap data to identify areas composed mainly of sandy sediment (Duncan 2016) and the OSPAR database of threatened and/or declining

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[5] http://jncc.defra.gov.uk/pdf/P20161206_UK_Sandbanks_v2_Method_1_1.pdf

habitats to identify areas vegetated with *Zostera marina* and/or free living species of the Corallinaceae family (Ellwood & Duncan 2015). The total surface area of this feature in UK waters is currently estimated to be 17,090 km^2. This is based on current best available evidence that includes modelled data with lower confidence and may therefore be an under-estimate.[6]

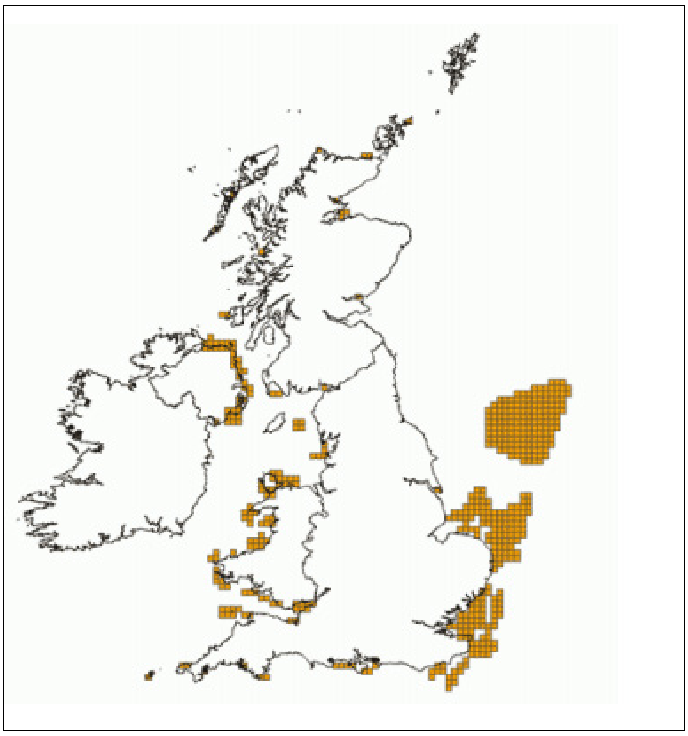

Figure 8.1.5 *The potential area of shallow sublittoral sandbanks in UK waters (from: Third Report by the United Kingdom under Article 17. H1110 Sandbanks which are slightly covered by sea water all the time)*[7]

Conclusions

The development and maintenance of sandbanks is strongly dependent on sediment supply, and hydrographic conditions. Admiralty Charts and other sources reveal that there has been movement and change in the size/area covered by some sandbanks over both long (centuries) and short (decades) time scales. There are examples of reductions in area as well as situations where there are gains or no detectable change. Considering the dynamic character of the habitat and without any clear evidence for strong declines in area, the FRA for this habitat can be taken as equivalent to current value. However, as further data becomes available this figure should be updated.

FRA (UK) = CV (17,090 km^2).

Step 2.2 - FRR assessment

The range of shallow sandbank habitat in UK waters is estimated to be 103,943 km^2 (Figure 8.1.6). This was calculated by identifying areas of sandy sediment in less than 60 m of water depth that are connected to areas of sandy sediment in less than 20 m of water. Mapped data was created by combining existing data of sandbanks already mapped with an analysis of bathymetric depth, slope and aspect, and sediment data across UK waters. The figure, which was provided in the 2006-2012 Article 17 report for the Habitats Directive, is considered likely to be an over estimate at this time, as it does not exclude flat and gently sloping sand.

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[6] UK Article 17 report for 2006- 2012. http://cdr.eionet.europa.eu/Converters/run_conversion?file=gb/eu/art17/envukl4fa/GB_habitats_reports-13925-152535.xml&conv=350&source=remote#1110

[7] http://jncc.defra.gov.uk/pdf/Article17Consult_20131010/h1110_UK.pdf

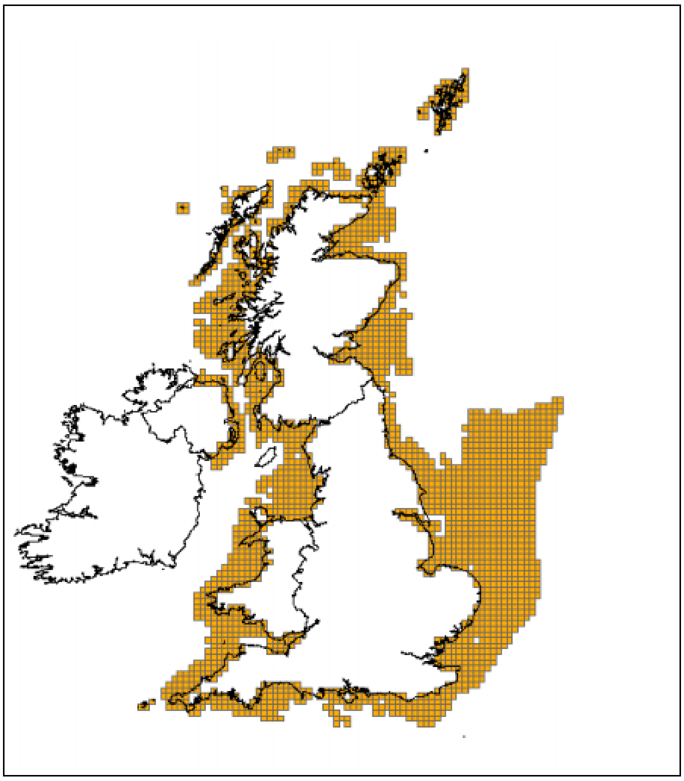

Figure 8.1.6 *The potential range of shallow sublittoral sandbanks in UK waters (from: Third Report by the United Kingdom under Article 17. H1110 Sandbanks which are slightly covered by sea water all the time)*[3]

Conclusions

Sandbank features occur throughout the shallow sublittoral in areas where there is an adequate sediment supply and favourable hydrographic conditions. They can be highly dynamic, moving many kilometres over relatively short periods of time but there is no evidence of any significant alteration in their range in either the historic (50 years) or recent (10 years) past in UK waters. The FRV for this habitat can therefore be taken as equivalent to current value.

FRR (UK) = CV (103,943 km^2).

References

American Geological Institute (1974). Dictionary of Geological Terms. National Academy of Sciences. Anchor Press/Doubleday, New York.

Boedeker, D., J.C. Krause & H. von Nordheim (2006). Interpretation, identification and ecological assessment of the Natura 2000 habitats "sandbank" and "reef". In: Von Nordheim, H. *et al.* (eds). Progress in Marine Conservation in Europe. Natura 2000 Sites in German Offshore waters. Springer, Berlin, pgs 47-64.

Brière, C., P.C. Roos, E. Garel & S.J.M.H. Hulscher (2010). Modelling the morphodynamics of the Kwinte Bank, subject to sand extraction. Journal of Coastal Research 51: 117-126.

Burningham, H. & J.R. French (2008). Historical changes in the seabed of the greater Thames estuary. The Crown Estate, 54 pp.

Camphuysen, K., B. Scott & S. Wanless (2011). Distribution and foraging interactions of seabirds and marine mammals in the North Sea: a metapopulation analysis. http://www.abdn.ac.uk/staffpages/uploads/nhi635/ZSLpaper-kees.pdf.

Dyer, K.R. & D.A. Huntley (1999). The origin, classification and modelling of sandbanks and ridges. Continental Shelf Research 19: 1285-1330.

Duncan, G. (2016). Method for creating version 2 of the UK composite map of Annex 1 sandbanks slightly covered by seawater all of the time. http://jncc.defra.gov.uk/PDF/P20161206_UK_Sandbanks_v2_Method_1_0.pdf [Accessed 5.5.17]

Ellwood, H. & G. Duncan, G. (2015). Creating a composite OSPAR threatened and/or declining habitat map for the UK (version 1.0). http://jncc.defra.gov.uk/PDF/R20150610 compositeOSPARHabitatMapMethod_v1.0.pdf [Accessed 5.5.17]

European Commission (2013). Interpretation Manual of European Union Habitats. EUR 28. April 2013. European Commission, DG Environment, Brussels, 146 pp.

JNCC, Joint Nature Conservation Committee (2017). Annex 1 Sandbanks in offshore waters. http://jncc.defra.gov.uk/page-1452. [Accessed 28.04.17]

Jones, S. (2001, ed.). Implementation of the EU Habitats Directive Offshore: Natura 2000 sites for reefs and submerged sandbanks. Volume II: Northeast Atlantic and North Sea. WWF UK, Surrey 92 pp.

Jones, E., B. McConnell, C. Sparling & J. Matthiopoulos (2013). Grey and harbour seal density maps. Report from the Sea Mammal Research Unit to Marine Scotland. http://www.scotland.gov.uk/Resource/0041/00416981.pdf

Kenyon, N.H. & B. Cooper (2005). Sand banks, sand transport and offshore windfarms. Report Kenyon MarineGeo & ABP Marine Environmental Researh Ltd., 106pp.

Kaiser, M.J., M. Bergmann, H. Hinz, M. Galanidi, R. Shucksmith, E.I.S. Rees, T. Darvyshire & K. Ramsay (2004). Demersal fish and epifauna associated with sandbank habitats. Estuarine, Coastal and Shelf Science 60: 445-456.

Klein, A. (2006). Identification of submarine banks in the North Sea and the Baltic Sea with the aid of TIN modelling. In: Von Nordheim, H. *et al.* (eds). Progress in Marine Conservation in Europe. Natura 2000 Sites in German Offshore waters. Springer, Berlin, pgs 97-110.

Kröncke, I. (1992). Macrofauna standing stock of the Dogger Bank. A comparison: III. 1950-54 versus 1985-87. A final summary. Helgolander Meeresunters. 46: 137-169.

Kröncke, I., H. Reiss, J. Eggleton *et al.* (2011). Changes in North Sea macrofauna communities and species distribution between 1986 and 2000. Estuarine, Coastal and Shelf Science 94: 1-15.

Lewis, M.J., S.P. Neill & A.J. Elliott (2015). Interannual variability of two offshore sand banks in a region of extreme tidal range. Journal of Coastal Research 31(2): 265-275.

Schratzberger, M. & P. Larcombe (2014). The role of the sedimentary regime in shaping the distribution of subtidal sandbank environments and the associated meiofaunal nematode communities: an example from the southern North Sea. PLOS one 9(10): e109445. http://journals.plos.org/plosone/article?id=10.1371/journal.pone.0109445

Skov, H., J. Durinck, M.F. Leopold & M.L. Tasker (1995). Important Bird Areas for seabirds in the North Sea, including the Channel and Kattegat. BirdLife International, Cambridge.

Skov, H., S. Heinanen, C.B. Thaxter, A.E. Williams, S. Lohier & A.N. Banks (2016). Real-time species distribution models for conservation and management of natural resources in marine environments. Marine Ecological Progress Series 542: 221-234.

Thompson, P.M. & D. Miller (1990). Summer Foraging Activity and Movements of Radio-Tagged Common Seals (Phoca vitulina.L). in the Moray Firth, Scotland. Journal of Applied Ecology 27(2): 492-501.

Vanaverbeke, J., T. Gheskiere & M. Vincx (2000). The meiobenthos of subtidal sandbanks on the Belgian Continental Shelf (Southern Bight of the North Sea). Estuarine, Coastal and Shelf Science 51: 637-649.

Velegrakis, A.F., M.B. Collins, G. Owrid, & A. Houghton (2001). Submerged sandbanks of the European Shelf Waters. In: Jones, S. (ed.). Implementation of the EU HABITATS Directive Offshore: Natura 2000 sites for reefs and submerged sandbanks. Volume II: Northeast Atlantic and North Sea. WWF UK Report, Surrey, pp. 44-85.

8.2 *Posidonia* beds (1120) in the Mediterranean with focus on Malta

Step 1.1 - Ecology of the habitat

Posidonia oceanica is a seagrass which forms extensive underwater meadows in water depths where sufficient light penetrates to support photosynthesis and growth. It is large (leaves can be up to 1m long during the height of summer) and slow growing, with wide and persistent rhizomes. Old leaves are shed throughout the year, but especially in the autumn when the canopy appears shorter and sparser (Díaz-Almela & Duarte 2008). Enhanced sedimentation, combined with vertical rhizome growth produces a characteristic "matte". This is a network of dead rhizomes with shell/organic debris and sediments which accumulative over centuries and can be several meters in height (Hemminga & Duarte 2000). *Posidonia* beds are important carbon sinks, stabilize sediment, protect coastline from erosion, support biodiversity and enhance living resources (Barbier *et al.* 2011).

This habitat can develop in areas where the seabed is rocky or sandy with the *Posidonia* plants using the substrate for anchorage and nutrient uptake. Muddy seabed is unsuitable because of difficulties with attachment and turbidity which limits plant growth. In sheltered bays *Posidonia* meadows may grow up to the surface whereas in more wave exposed conditions they usually start growing several meters below the surface (3-10m) (Díaz-Almela & Duarte 2008).

Posidonia beds are one of the main climax stages of Mediterranean coastal ecosystems. They harbour a highly diverse community, which varies according to depth, shoot density, adjacent communities, physico-chemical conditions and even historical events linked to larval recruitment (Díaz-Almela & Duarte 2008). Epiphytic community comprises micro and macro algae, sessile animals such as hydroids, bryozoans, and microscopic foraminifera. Within the matte and amongst the rhizomes are foraminifera, fan mussels, and algae adapted to low levels of light. Molluscs, sponges, echinoderms (sea urchins, brittle starts, feather stars and sea cucumbers) are also present moving to feed in the canopy at night. *Posidonia* beds act as nursery areas for many fish as well as supporting resident species including gobies, wrasse, and chromis. The endangered sea horse *Hippocampus hippocampus* is also present and the herbivorous green turtle *Chelonia mydas* feeds on the seagrass (Díaz-Almela & Duarte 2008).

Posidonia oceanica is endemic to the Mediterranean Sea and *Posidonia* beds have been identified as a priority habitat type for conservation in Annex 1 of the Habitats Directive.

A variety of parameters are used to monitor seagrass habitat quality. They include descriptors at an individual level (shoot phenology and biomass), at a population level (bed density and coverage) or at a community level (epiphytes on leaves). *Posidionia oceanica* is also used as a biotic index to evaluate the ecological status of coastal waters under the MSFD (e.g. Fernández-Torquemada *et al.* 2008; Montefalcone 2009).

This habitat type is present along the continental coastline and islands of all EU Mediterranean countries. The current distribution is shown in Figure 8.2.1 and the total area estimated at 12,247 km^2.[8]

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[8] A5.535 *Posidonia* beds in the Mediterranean infralittoral zone. European Red List of Habitats. http://forum.eionet.europa.eu/european-red-list-habitats/library/marine-habitats/mediterranean-sea/a5.535-posidonia-beds-mediterranean-infralittoral-zone.

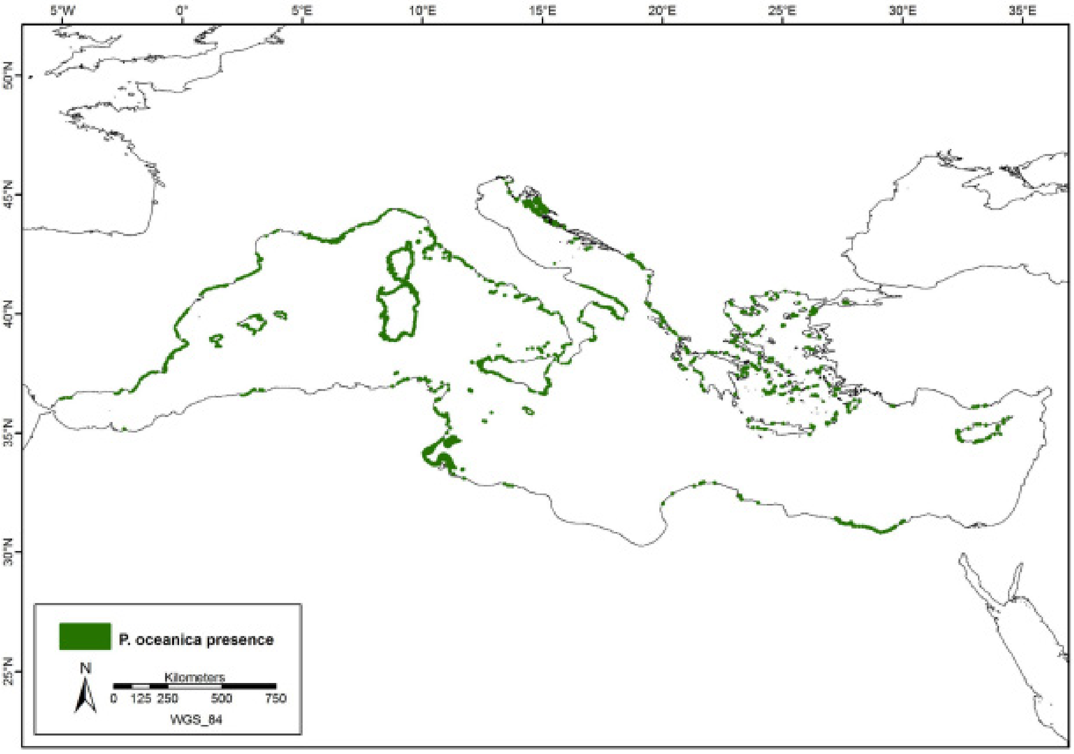


Figure 8.2.1 *Current distribution of Posidonia meadows (green areas) along the Mediterranean Sea coastline (from Telesca et al. 2015)*

Step 1.2 - Spatial scale of functioning

Posidonia beds are present in all the sub-basins of the Mediterranean Sea and there are slight morphological and genetic differences between *P. oceanica* meadows from different regions. In particular, there is a genetic cleavage between the Eastern and Western Mediterranean meadows which suggests that these meadows were temporally isolated from each other during last glaciations (Arnaud-Haond *et al.* 2007). Nevertheless, there are no clear geographical differences in meadow structure and function between the two basins, and the morph type differences disappear after some years of acclimatization when transplanted to another site (Meinesz *et al.* 1993).

Posidonia oceanica reproduces both sexually and asexually. Vegetative propagation by subterranean rhizomes appears to be very important for the physical and stability and resilience of meadows whereas sexual reproduction probably functions in long distance species dispersal and for genetic diversity (Procaccini *et al.* 1996; Buia & Mazzella 1991).

The status and condition of this habitat is determined by the characteristic species *Posidonia oceanica*. Girauld (1977) identified types of *Posidonia* bed based on shoot density and Pergent *et al.* (1995) developed this into a scale of habitat condition across different depth zones (see Figure 8.2.2). These studies were concerned with investigating the use of *Posidonia* as a potential descriptor of water quality rather than of the habitat itself. A population-based method for setting FRVs using *Posidonia oceanica* as the diagnostic species may therefore be an option, however a reference-based method using area of habitat is considered a more practical approach at current levels of information about this habitat throughout its distribution.

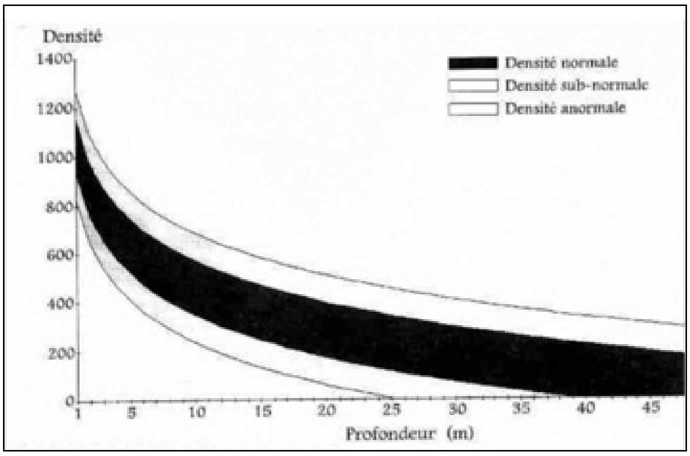


Figure 8.2.2 *Significance of the density of Posidonia beds as a function of depth (from Pergent et al. 1995)*

Conclusions

Posidonia beds develop under specific abiotic conditions. They are a climax community with species composition, structure, site conditions and spatial connectivity being criteria for good quality. This can be considered a mesohabitat and therefore best categorized as habitat type 2a.

The status and condition of this habitat is determined by the characteristic species Posidonia oceanica, however whilst a population-based method for setting FRVs may be possible in the future a reference based approach using area of habitat (combined with density data where available) is considered the more practical option at the present time. There are no clear differences across the Mediterranean basins, therefore it is recommended that assessment is carried out at a national level.

Step 1.3 - Historical perspective: what happened to the habitat?

Trends in *Posidonia* beds are revealed by analysis of a variety of types of data. Changes in the quantity of this habitat typically relies on records of presence/absence, areal extent, and density of plants. Trends in quality may be based on changes in parameters such as shoot density, and the abundance and diversity of associated species.

The first attempts at mapping *Posidonia oceanica* beds date back to the end of the 19th century although the first maps were not produced until the early 1970s in France and Italy. Efforts to evaluate the status of this habitat started in 1962 with most of the knowledge to date based on observations in the western Mediterranean.

There is evidence that *Posidonia oceanica* has been suffering widespread decline, at least in the north-west Mediterranean (Marbá *et al.* 1996; Pascualini *et al.* 1999; Boudouresque *et al.* 2000). A review of the literature on *Posidonia* beds over the period 1842-2009 (Marbà *et al.* 2014) concluded that two thirds of the 519 meadows assessed had experience declines in areal extent, cover and/or shoot density (Figure 8.2.3). There were also changes in the depth limits of meadows over the last century with most of the meadows evaluated regressing towards deeper waters. The percentage of cover and shoot density in areas where meadow was not lost also exhibited changes during the last decides. In half of the cases there was a reduction, 44% were in a steady state, and one meadow showed an increase in cover. Overall there was a net decline of cover at on average -1.22% yr^{-1}. The shoot density in 43% of meadows examined had thinned.

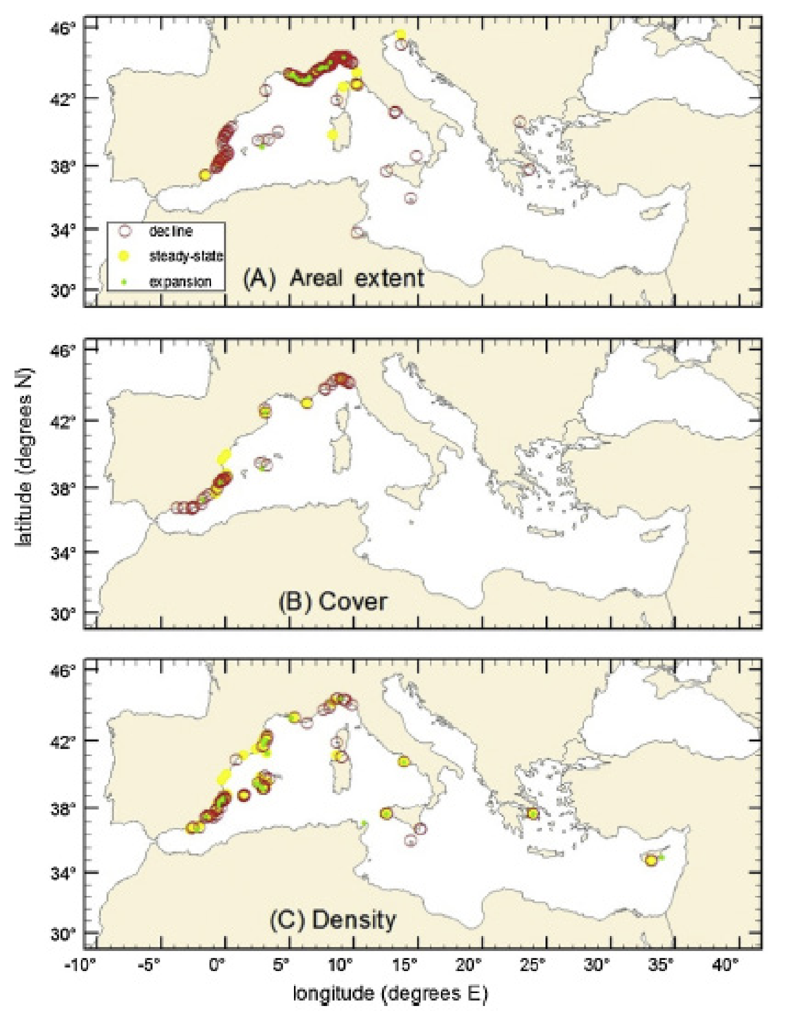


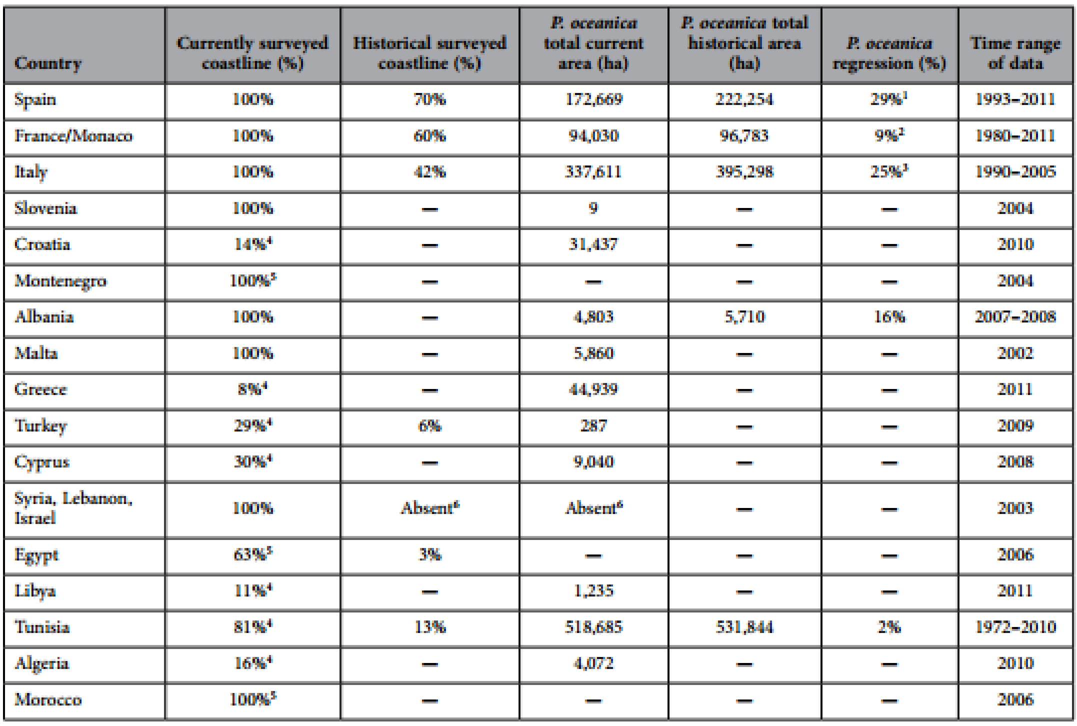
Figure 8.2.3 *Distribution of all Posidonia oceanica meadows that have been reported to be declining, in steady-state, and expanding determined by assessing meadow areal extent, cover and shoot density (from Marbà et al. 2014)*

Regression of meadows is a generalised phenomenon in the Mediterranean Sea, even though some exceptions exist (e.g. Corsica, parts of the Sardinian coastline and the Valencia region in Spain). There are also some uncertainties because of changes in mapping techniques (Bonacorsi *et al.* 2013). During the twentieth century and especially in the 1950 s, *P. oceanica* meadows have considerably regressed, mainly near large urban developments and ports such as Barcelona, Marseille, Toulon, Genoa, Trieste, Alexandria and Gabès. Telesca *et al.* (2015) have collated data on current and known historical areas and percentage regression over various time periods (Table 8.2.1 and Figure 8.2.4).

Comparison of current distribution maps with available historical ones allows assessment of the changes undergone by meadows over time (Figure 8.2.4). In areas for which historical data were available, the estimated regression of *Posidonia* meadows in the EU 28 was 34% over the last 50 years. There has also been a decline in the overall quality, estimated to be at least a slight decline affecting more than 30% of the extent of the habitat over the same period (Gubbay *et al.* 2016).

Because of the requirements of seagrass for adequate light and sediment conditions, they are affected by disturbances that modify water and sediment quality. The shallow coastal environment occupied by seagrass beds is also particularly prone to physical disturbance, whether by waves or turbulence associated with stormy weather.

***Table 8.2.1** Lengths of coastline with the known current and historical presence of Posidonia oceanica, the percentage of regression and the time range of data (from Telesca et al. 2015; see this publication for details and footnotes)*



Country	Currently surveyed coastline (%)	Historical surveyed coastline (%)	*P. oceanica* total current area (ha)	*P. oceanica* total historical area (ha)	*P. oceanica* regression (%)	Time range of data
Spain	100%	70%	172,669	222,254	29%[1]	1993–2011
France/Monaco	100%	60%	94,030	96,783	9%[2]	1980–2011
Italy	100%	42%	337,611	395,298	25%[3]	1990–2005
Slovenia	100%	—	9	—	—	2004
Croatia	14%[4]	—	31,437	—	—	2010
Montenegro	100%[5]	—	—	—	—	2004
Albania	100%	—	4,803	5,710	16%	2007–2008
Malta	100%	—	5,860	—	—	2002
Greece	8%[4]	—	44,939	—	—	2011
Turkey	29%[4]	6%	287	—	—	2009
Cyprus	30%[4]	—	9,040	—	—	2008
Syria, Lebanon, Israel	100%	Absent[6]	Absent[6]	—	—	2003
Egypt	63%[5]	3%	—	—	—	2006
Libya	11%[4]	—	1,235	—	—	2011
Tunisia	81%[4]	13%	518,685	531,844	2%	1972–2010
Algeria	16%[4]	—	4,072	—	—	2010
Morocco	100%[5]	—	—	—	—	2006

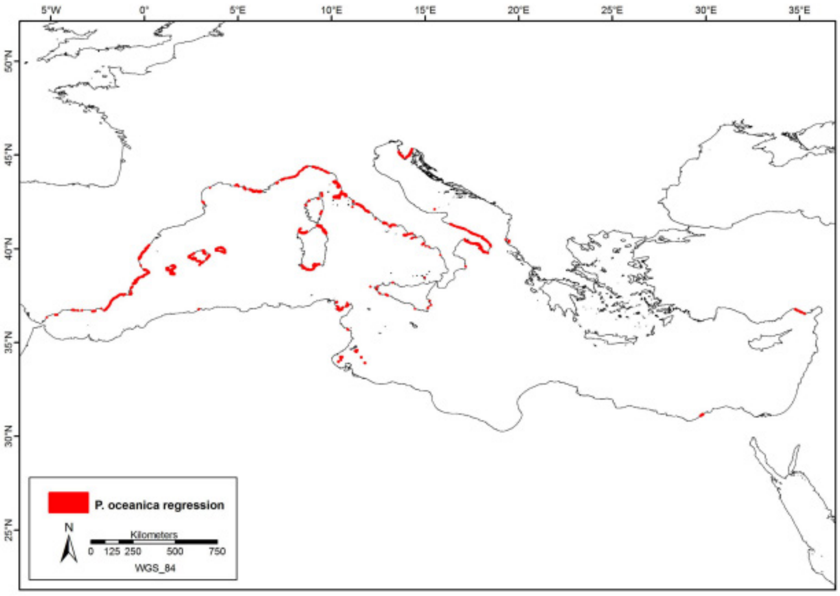


Figure 8.2.4 *Coastline with regression of Posidonia oceanica meadows (from Telesca et al. 2015)*

Anthropogenic pressure on the Mediterranean coastal zone increased rapidly during the second half of the 20th century with widespread seagrass loss directly attributed to human activity. The main pressures on this habitat have been mechanical damage (by dredging, fishing, and anchoring), eutrophication, aquaculture, siltation, effects of coastal constructions, and food web alterations. Indirect effects associated with human activity include negative effects of climate change particularly through impacts of elevated seawater temperature on shoot survival (Marbà & Duarte 2010; see Figure 8.2.5) but also erosion by rising sea level assuming growth cannot keep pace with changing sea level, increased storms, increased ultraviolet irradiance, as well as from natural causes, such as cyclones and floods. In all regions, the environmental effects of excess nutrients or sediments are the most common and significant causes of seagrass decline, and result in small to very large areas of seagrass being lost (Orth *et al.* 2006). Invasive macroalgae such as *Caulerpa taxifolia* can grow on *Posidonia* rhizomes and sand and their proliferation is believed could accelerate the decline of the meadow.

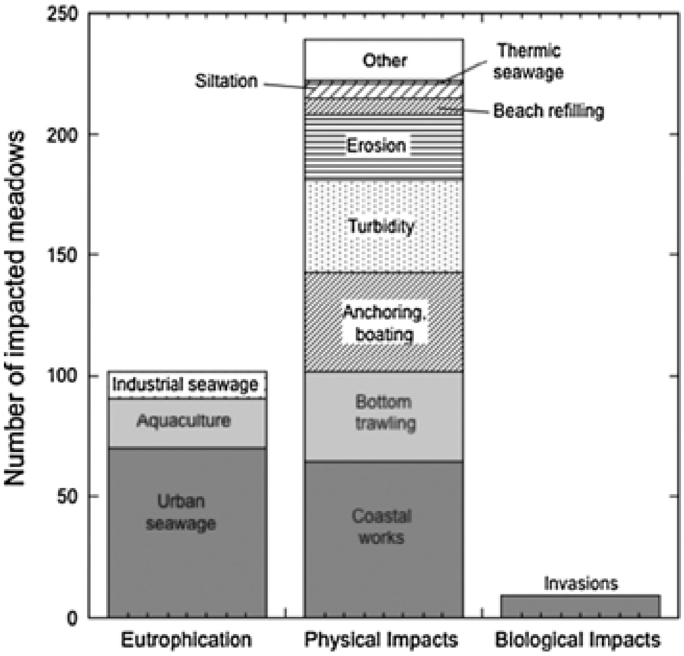


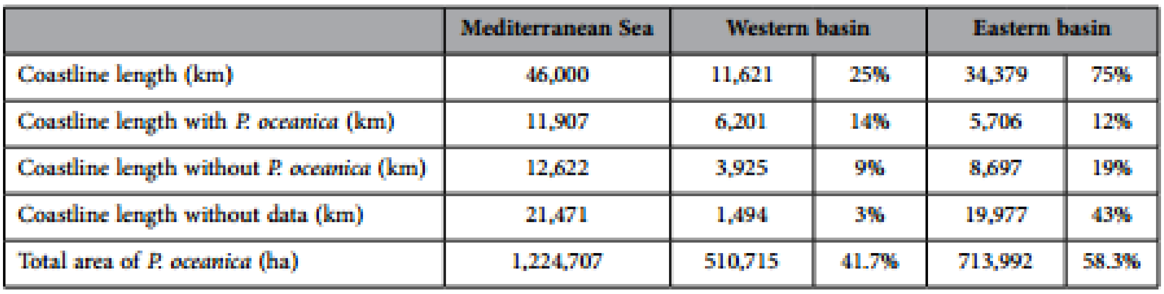
Figure 8.2.5 *Pressures identified as causes of Posidonia oceanica declines. The graph shows the number of meadows impacted by each pressure (from Marbà et al. 2014)*

Whereas local human impacts are often evident as causes for the decline of *Posidonia* beds, this is not always the case: significant correlation between decadal growth and climate variation (Marbá & Duarte 1997) points to climate change effects and possibly indirect human impacts on the observed decline. These observations suggest that seagrass decline and, possibly expansion, may be triggered by large-scale climate change. If so, rapidly changing climatic patterns may lead to major changes in global seagrass cover in the future (Duarte 2002).

The biological characteristics of *Posidonia* with rare sexual reproduction and slow horizontal growth of rhizome edges, do not favour rapid recolonization of degraded beds or formation of new beds. Transplantation has been attempted to restock and restore *Posidonia* beds, but whilst the survival rate of cuttings can be very good, recolonization is always slow. More than 25 years after the first transplanting was done, there are still no true meadows reconstituted from transplants (Boudouresque *et al.* 2012).

In general, the area of this habitat is known to have decreased during the 20th century and considered to have declined in both quality and quantity across the Mediterranean. However, there are some areas and time periods where there has been some recovery, most likely associated with improvements in water quality (Boudouresque *et al.* 2000), and occasional stability or small increases (Telesca *et al.* 2015). One estimate based on coastal seafloor between 0-40 m is that this habitat could have covered about 50 000 km^2 in the Mediterranean in the past (Bethoux & Copin-Montégut 1986). The total known current area of *Posidonia* beds has been calculated as just over 1.2 million ha (Table 8.2.2; Telesca *et al.* 2015).

Table 8.2.2 *Spatial extent of Posidonia oceanica meadows across the Mediterranean Sea (from Telesca et al. 2015)*



	Mediterranean Sea	Western basin		Eastern basin	
Coastline length (km)	46,000	11,621	25%	34,379	75%
Coastline length with *P. oceanica* (km)	11,907	6,201	14%	5,706	12%
Coastline length without *P. oceanica* (km)	12,622	3,925	9%	8,697	19%
Coastline length without data (km)	21,471	1,494	3%	19,977	43%
Total area of *P. oceanica* (ha)	1,224,707	510,715	41.7%	713,992	58.3%

Step 1.4 - Analysis of distribution and trends (Malta)

In the Maltese Islands, Posidonia meadows occur as two main subtypes; the continuous meadows and the reticulate or non-continuous meadows (Borg & Schembri, 2002) The earliest mapping of the spatial extent of *Posidonia* beds around the Maltese islands was undertaken in 1994 using SCUBA diving techniques (Borg & Schembri 1995). This reported particularly dense and healthy meadows in the Malta-Comino and Comino-Gozo channels as well as patchy stands in shallow waters and dense meadows down to 25-30m on soft sediment.

Both diving and geophysical survey techniques were used to prepare baseline maps in 2002 (Borg & Schembri 1995; Geological Assistance & Services 2003). Both surveys revealed that the habitat covers large areas of suitable shallow seabed around the Maltese islands. It is present along the north-eastern to south-eastern coast of Malta with the densest meadows to the east of mainland Malta. *Posidonia* beds are also common around Gozo, along its north-eastern coast, as well as in the channels between Gozo and Comino, and between Comino and Malta (Geological Assistance & Services 2003). The habitat has developed on rocky shelf, gravel, sand, and matte structures. The beds are characterized by healthy shoots forming extensive bands over considerable areas of matte (Borg *et al.* 2013). One of the deepest records for *Posidonia* in the Mediterranean is off the south-western coast of Comino Island where a sparse meadow is present at a depth of around 43m (Borg *et al.* 2006). The baseline survey undertaken in 2002 reported the estimated area of the major *Posidonia* beds as in Table 8.2.3, with many other smaller patches also present (Geological Assistance & Services 2003).

Table 8.2.3 *Main areas of Posidonia beds around the Maltese islands (from Geological Assistance & Services 2003). This excludes patches and scattered areas of Posidonia*

Location	Area (km^2)
Gozo island	3.9
Gozo-Comino Islands	2.1
Comino-Malta Islands	27
Malta	7.4

This data has since been pooled with further information from four locations where *Posidonia* beds were studied in detailed confirming the general distribution around the Maltese islands (Borg *et al.* 2009) given in Figure 8.2.6.

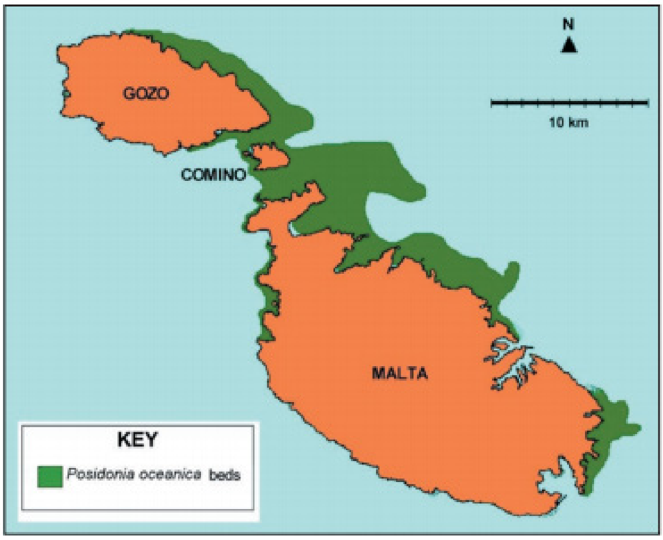


Figure 8.2.6 *Small scale map showing the large-scale distribution of Posidonia oceanica around the Maltese Islands (from Borg et al. 2009)*

There is a lack of data on the historical extent of this habitat but it is known to have regressed or been lost from some locations in recent decades. Reports of some declines were published in the Malta Red Data book in 1989 (Schembri & Sultana 1989). In the 1990s Borg & Schembri (1995) noted that frequent anchoring was probably the main factor causing significant damage to the extensive meadows originally present in Mellieha Bay. They also reported the regression and replacement of some *Posidonia* beds by pollution tolerant benthic communities in several bays and creeks (e.g. Marsamxett and Grand Harbour), and loss of *Posidonia* beds associated with offshore fish farming operations at Mistra Bay and Mellieha Bay, and by beach recharge at Pretty Bay, Birzebbuga. Around a decade later, Borg *et al.* (2009) note that some fragmentation has almost certainly occurred in some areas, particularly in the vicinity of the Valletta harbours and off Xghajara where the largest sewage outfall is located. Elevated nutrient loading and more intense coastal use is also likely to have resulted in more stressed beds off the southern coasts compared to off the northern coasts.

Besides, changes in *Posidonia* have been reported from a site in St. Paul’s Bay on the north-west coast of Malta associated with fish farming. Plants directly below the cages were lost soon after operations started. Those 10 m away from the perimeter of the cages retained their vitality but with reduced shoot density and with heavy epiphytic growth on the leaves (Dimech *et al.* 2002). In the Malta-Comino Channel part of a *Posidonia* bed has been lost from underneath and within a 30m wide band around offshore fish cages that were operating between 1995-2000. This is likely to have been the result of the combined effect of shading and enrichment of the sediment and water column by waste from the farm. There were also lower shoot densities than reported for the same meadow prior to initiation of the fishing arming activities, but also some indication of potential recovery following. Comparative data on shoot density, shoot area and shoot biomass at 19 sites between 1999 and 2004 determined that the majority were in good condition (Debono & Borg 2006).

Conclusions

Posidonia beds are known to have been much more widespread and abundant in the Mediterranean with declines in extent and quality associated with human activity. These trends are well reported from the second part of the 20th century onwards.
Posidonia beds are known to have been impacted by human activity. The main pressures identified are mechanical damage, eutrophication, aquaculture, siltation, effects of coastal constructions, and food web alterations. Indirect effects associated with human activity include negative effects of climate change and this is considered likely to increase in the future. Removing threats is essential for recovery and whilst transplantation may assist restoration of some Posidonia beds, this will be a slow process and success is not guaranteed.

Historical data on the extent and abundance of this habitat around the Maltese islands is lacking. There are also limited data from the recent past (50 years) apart from some information from the 1970s relating to specific locations. Relevant records for assessment and trends are mostly from the 1990s onwards. A comprehensive survey of this habitat around Malta was undertaken in 2002. Recent observations indicate some regression and loss of habitat in the recent past although the scale cannot be quantified. The Habitats Directive came into force in Malta in 2004.
Currently there are dense, healthy Posidonia meadows covering large areas of seabed off the Maltese islands and extending to considerable depths, although there have been some losses. Where it is present the habitat is mostly in favorable condition and covering large areas of suitable shallow seabed. It is therefore likely that in this case FRV can be taken as equivalent to DV (‘directive value’).

Step 2.1 - FRA assessment (Malta)
In Malta several examples of decline in area and quality have been reported, but overall these impacts are small. Dense, healthy *Posidonia* meadows cover large areas of seabed off the Maltese islands, extending to considerable depths. Where it is present the habitat is mostly in favorable condition and covering large areas of suitable shallow seabed. The FRV is taken as at least the value of the baseline study in 2003 shortly after the Habitats Directive came into force, a figure which does not take account of patchy and scattered areas of this habitat. Article 17 reporting for the period 2001-2006 gives the area of *Posidonia* beds in Malta as 168 km^2. Reporting for the 2007-2012 gives a figure of 179 km^2.

Conclusions

FRA = CV (179 km^2).

Step 2.2 - FRR assessment (Malta)
All along the Mediterranean coast, except in the northern Adriatic, the seagrass *Posidonia oceanica* lies between 0 and 35 m depth, with a mean sea floor occupation of 50% (Bethoux & Copin-Montegut 1986). Whilst the area of this habitat is considered to be decreasing at a Mediterranean scale it is unclear if there has been any reduction in horizontal or depth range.

A 2002 baseline survey of this habitat around the Maltese islands revealed that it was widespread across suitable areas (Geological Assistance & Services 2003). *Posidonia* beds were present along the north-eastern to south-eastern coast of Malta with the densest meadows to the east of mainland Malta. It is also widely present along the north-eastern coast and in two areas to the west of Gozo, in the channels between Gozo and Comino, and between Comino and Malta. The only break in occurrence on the eastern coasts are between the Valletta harbours and Xghajra. The habitat is much less extensive on the western coasts, mainly between Mgarr ix-Xini on the southern coast of Gozo and Fomm ir-RIh but with some meadows and patches in the smaller inlets (Borg *et al.* 2009). There is no historical data on the range of *Posidonia* beds around the Maltese islands but its widespread current distribution, including to considerable depth suggests there is unlikely to have been any significant reduction in range in recent years. The FRR is sufficient large to include the FRA, and therefore current value is considered as the FRR[9].

Conclusions

FRR = CV (1100 km^2 based on the map included in the 2012-reporting).

References

Arnaud-Haond, S., E. Díaz Almela, S. Teixeira, F. Alberto, C.M. Duarte & E.A. Serrao (2007). Vicariance patterns in the Mediterranean sea: East-West cleavage and low dispersal in the endemic seagrass *Posidonia oceanica*. Journal of Biogeography 34: 963-976.

Barbier, E.B., S.D. Hacker, C. Kennedy, E.W. Koch, A.C. Stier & B.R. Sillman (2011). The value of estuarine and coastal ecosystem services. Ecological Monographs 81: 169-193.

Bethoux, J.P. & C. Copin-Montégut (1986). Biological fixation of atmospheric nitrogen in the Mediterranean Sea. Limnology and Oceanography 31: 1353–8.

Bonacorsi, M., C. Pergent-Martini, N. Breand & G. Pergtent (2013). Is Posidonia oceanica regression a general feature in the Mediterranean Sea? Mediterranean Marine Science 14(1): 193-203.

Borg, J.A., & Schembri, P.J. (1995). The state of Posidonia oceanica (L.) Delile meadow in the Maltese Islands (Central Mediterranean). Rapport du Congrès de la Commission Internationale pour l'Exploration Scientifique de la Mer Méditerranée 34: 123.

Borg, J.A., & Schembri, P.J. (2002). Alignment of marine habitat data of the Maltese Islands to conform to the requirements of the EU Habitats Directive (Council Directive 92/43/EEC). Report Commissioned by the Malta Environment & Planning Authority. Malta. Independent Consultants; 136pp +Figs 1-23.

Borg, J.A., M.A. Micallef, M.A. & P.J. Schembri (2006). Spatio-temporal variation in the structure of a deep water Posidonia oceanica meadow assessed using non-destructive techniques. Marine Ecology 27: 320-327.

Borg, J.A., A.A. Rowden, M.J. Attrill, P.J. Schembri, & M.B. Jones (2009). Occurrence and distribution of different bed types of seagrass Posidonia oceanica around the Maltese Islands. Mediterranean Marine Science 10(2): 45-61.

Borg, J.A., L. Knittweis & P.J. Schembri (2013). Compilation of an interpretation manual for marine habitats within the 25 nm fisheries management zone around the Republic of Malta. EcoServ Ltd., 219 pp.

Boudouresque, C.F., E. Charbonel, A. Meinesz, G. Pergent, C. Pergent-Martini, G. Cadiou, M.C. Bertrandy, P. Foret, M. Ragazzi, V. Rico-Rainmondino (2000). A monitoring network based on the seagrass Posidonia oceanica in the Northwester Mediterranean Sea. Biologia Marina Mediterranea 7(2): 328-331.

Boudouresque, C.F., G. Bernard, P. Bonhomme, E. Charbonnel, G. Diviacco, A. Meinesz, G. Pergent, C. Pergent-Martini, S. Ruitton & L. Tunesi (2012). Protection *and conservation of Posidonia oceanica meadows*. RAC/SPA & RAMOGE, 202 pg.

Buia, M.C. & L. Mazzella (1991). Reproductive phenology of the Mediterranean seagrasses Posidonia oceanica (L.) Delile, Cymodocea nodosa (Ucria) Aschers., and Zostera noltii Hornem. Aquatic Botany 40: 343-362.

Debono, S. & Borg, J.A. (2006). Use of Posidonia oceanica as a bioindicator of ecological status for Maltese coastal waters. Biologia Marina Mediterranea 13(4): 206-209.

Díaz-Almela E. & Duarte, C.M. 2008. Management of Natura 2000 habitats. 1120 * Posidonia beds (Posidonion oceanicae). Report, European Commission.

Dimech, M. Borg, J.A. & Schembri, P.J. (2002). Changes in the structure of a Posidonia oceanica meadow and in the diversity of associated decapod mollusc and echinoderm assemblages,

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[9] Article 17 reporting on this habitat gives a surface area of the range as 172 km^2 in 2007 and 185 km^2 in 2012. However, the map included in the 2012-reporting indicates 11 10x10 km squares (1100 km^2). http://art17.eionet.europa.eu/article17/reports2012/habitat/summary/?period=3&group=Coastal+habitats&subject=1120&region=

resulting from inputs of waste from a marine fish farm (Malta, Central Mediterranean). Bull.Mar.Sci. 71(3): 1309-1321.

Duarte, C.M. (2002). The future of seagrass meadows. Environmental Conservation 29(2): 192-206.

Fernández-Torquemada, Y., M. Díaz-Valdés, F. Colilla, B. Luna, J. Luis Sánchez-Lizaso & A.A. Ramos-Espla (2008). Descriptors from Posidonia oceanica (L.) Delile meadows in coastal waters of Valencia, Spain, in the context of the EU Water Framework Directive. ICES Journal of Marine Science 65: 1492-1497.

Geological Assistance & Services (2003). Baseline survey of the extent and character of Posidonia oceanica (L.) Delile meadows in the territorial waters of the Maltese Islands. Final Report. https://www.mepa.org.mt/SOER_Documents/Posidonia_survey.pdf

Giraud, G. (1977). Contribution à la description et à la phénologie quantitative des herbierd de Posidonia oceanica (l.) Delile en Méditerranée. Rapport du Congrès de la Commission Internationale pour l'Exploration Scientifique de la Mer Méditerranée 24(4): 127-130.

Gubbay, S., N. Sanders, T. Haynes, J.A.M. Janssen, J.R. Rodwell, A. Nieto, M. García Criado, S. Beal, J. Borg, M. Kennedy, D. Micu, M. Otero, G. Saunders, M. Calix (2016). European Red List of Habitats. Part 1. Marine habitats. European Commission, DG Environment, Brussels, 52 pp. http://ec.europa.eu/environment/nature/knowledge/pdf/Marine_EU_red_list_report.pdf

Guillén, J.E., J.L. Sánchez Lizaso, S. Jiménez, J. Martínez, A. Codina, M. Montero, A. Triviño, G. Soler, J. Jacobo Zubcoff (2013). Evolution of Posidonia oceanica seagrass meadows and its implications for management. Journal of Sea Research 83: 65-71.

Hemminga, M.A. & C.M. Duarte (2000). Seagrass Ecology. Cambridge University Press, Cambridge.

Marbà, N., E. Díaz-Almela & C.M. Duarte (2014). Mediterranean seagrass (Posidonia oceanica) loss between 1842 and 2009. Biological Conservation 176: 183-190.

Marbá, N. & C.M. Duarte (1997). Interannual changes in seagrass (Posidonia oceanica) growth and environmental change in the Spanish Mediterranean littoral. Limnology and Oceanography 42: 800-810.

Marbá, N. & C.M. Duarte (2010). Mediterranean warming triggers seagrass (Posidonia oceanica) shoot mortality. Global Change Biology 16(8): 2366-2375. doi: 10.1111/j.1365-2486.2009.02130.x

Meinesz, A., Caye, G., Loquès, F., & Molenaar, H. Polymorphism and development of Posidonia oceanica transplanted from different parts of the Mediterranean into the National Park of Port-Cros. Bot.Mar. 36:209-216.(1993).

Montefalcone, M. 2009. Ecosystem health assessment using the Mediterranean seagrass Posidonia oceanica: A review. Ecosystem Indicators 9(4): 595-604.

Orth, R.J. T.J.B. Carruthers, W.C. Dennison, C.M. Duarte, J.W. Fourqurean, K.L. Heck jr., A.R. Hughes, G.A. Kendrick, W.J. Kenworthy, S. Olyarnik, F.T. Short, M. Waycott & S.L. Williams (2006). A global crisis for seagrass ecosystems. BioScience 56(12): 987-996.

Pergent, G., Pergent-Martini, C & Boudouresque, C-F. (1995). Utilisation de l'herbier a Posidonia oceanica comme indicateur biologique de la qualité due milieu littoral en Méditerranée état des connaissances. Mésogée 54: 3-27.

Pergent-Martini, C., V. Leoni, V. Pasqualini *et al.* (2005). Descriptors of Posidonia oceanica Meadows: Use and Application. Ecological Indicators 5: 213-230.

Pascualini, V., C. Pergent-Martin & G. Pergent (1999). Environmental impacts identification along Corsian coasts (Mediterranean Sea) using image processing. Aquatic Botany 65: 311-320.

Procaccini, G., Alberte, R.S. & Mazzella, L. 1995. Genetic structure of the seagrass Posidonia oceanica in the Western Mediterranean: ecological implications. Marine Ecological Progress Series 140: 153-160.

Schembri, P.J. & J. Sultana (1989, eds). Red Data Book for the Maltese Islands. Department of Information, Malta, viii + 142pp. + 8 plates.

Telesca, L., A. Belluscio, A. Criscoli, A., G. Ardizzone, E.T. Apostolaki, S. Fraschetti, M. Gristina, L. Knittweis, C.S. Martin, G. Pergent, A. Alagna, F. Badalamenti, G. Garofalo, V. Gerakaris, M.L. Pace, C. Pergent-Martini & M. Salomidi (2015). Seagrass meadows (Posidonia oceanica) distribution and trajectories of change. Scientific Reports 5: 12505. doi: 10.1038/srep12505 (2015).

9 Terrestrial habitats

9.1 Arborescent matorral with *Laurus nobilis* (5230*) in Italy

Emiliano Agrillo & Fabio Attorre

Step 1.1 - Ecology of the habitat

The habitat type 5230* refers to woodlands dominated by *Laurus nobilis* in the tree layer. It occurs in gorges or ravines where the topography offers a sheltered environment for the species against summer aridity and heat. Furthermore, the habitat is found in forests where the topography suits the thermo-hygrophilous requirements of laurel (warmth loving but drought sensitive). The habitat does not show a preference for substrate.

In Europe the *Lauraceae* family is represented by the the genus *Laurus* only. Traditionally, two species are recognized in the literature: *Laurus nobilis* L., with glabrous twig leaves, and *L. azorica* (Seub.) Franco with pubescent twig leaves (Arroyo-García *et al.*, 2001). However, the idea of the existence of only one species with considerable leaf plasticity is supported as well. *Laurus nobilis* is distributed along the three Mediterranean peninsulas (Iberian, Italian and Balkan), in Turkey, Israel, along the southern coast of the Black Sea and in the Atlantic coast of Portugal, Spain and France. Instead *L. azorica* is found in the Macaronesian region and in Morocco.

The requirement for shaded environments and preference for warm and moist climates of *Laurus nobilis* apparently haven’t been changed for the last 3 Myr. *Laurus* leaves can tolerate frost up to -10°C. Other physiological features such as low transpiration rate and a high capacity to recover from water shortage prevent water stress. Vegetative reproduction plays a great role in *L. nobilis* persistence, even in late successional stages where reproduction by seeds tends to become more prominent. The high capability of resprouting of *L. nobilis* from the *lignotuber* favoured habitat resilience to coppicing and fire events during the last centuries of human impact. Since pollen and seeds are dispersed with a water content of more than 30% and remain viable for only a few days, they are not well-adapted to long-distance disersal. Additionally, it is suggested that *Laurus* depends on just a few (bird) species for seed dispersal, at least outside Macaronesia (with endemic pigeons specialized on laurel species).

The vegetation with laurel can be very heterogeneous in floristic composition and structure. Frequently, laurel occurs as scattered shrubs in different woodland or in the mantel. Rarely *L. nobilis* can reach the tree layer or become dominant (as habitat type). Laurel often occurs as an element of azonal hygrophilous riparian woods, but it is also found in evergreen forest with dominance of *Quercus ilex* or in xerophytic rocky sites with *Fraxinus ornus*. Laurel participates also in sub-mediterranean *Q. pubescens s.l.* forests . In ravines *L. nobilis* can occur with mesophitic species such as *Fagus sylvatica*, *Carpinus betulus* or *Quercus robur*. This criterion allows it to be distinguished from 9340 “*Quercus ilex* and *Q. rotundifolia* forests”, where *L. nobilis* occurs as an associate species, or from 5310 “*Laurus nobilis* thickets”, where *L. nobilis* exhibits a shrub growth form. Nevertheless, others tree species, from the surrounding communities, can occur in the canopy (e.g. *Quercus ilex*, *Q. pubescens*, *Q. petraea*, *Carpinus betulus* or *Ostrya carpinifolia*). The understory is species-poor, due to the dense canopy of the evergreen dominant species. The most frequent species are lianas (e.g. *Hedera helix*, *Smilax aspera*, *Rubia peregrine, Asparagus acutifolius*) or evergreen shrubs (e.g. *Ruscus aculeatus*, *Viburnum tinus*, *Phillyrea latifolia*). The herb species composition is exceptionally poor, and often coming from nearest communities (e.g. *Cyclamen repandum*, *C. hederifolium*). In a condition of high level of atmospheric humidity, such as in deep gorges, the presence of ferns could confirm the good status of the habitat (e.g. *Asplenium onopteris*, *Polystichum setiferum*, *Polypodium cambricum*). The variation of habitat species composition along its altitudinal range (0-800 m.a.s.l.)

and geographical regions is due to the variation of the surrounding vegetation. Nevertheless, the most frequent species associated with this habitat do not vary.

The most favourable structure for the habitat 5230* is a well-developed and dense canopy, which permits the maintenance of a sheltered environment and the coexistence with few other evergreen species in the understory, enriched by ferns occurring in gorges or ravines. Juveniles and seedlings of these species, especially *L. nobilis,* have to be the most abundant in the rejuvenation layer. The availability of decomposing dead wood in different decay stages increases habitat heterogeneity and creates a variety of niches that sustain forest biodiversity.

Step 1.2 - Spatial scale of functioning

Within forest ecosystems, *L. nobilis* thickets are micro-forests with a punctual, or linear, scattered distribution due to their particular abiotic requirements. Thus, habitat heterogeneity and dynamical stages are determined mostly by surrounding ecosystems. Since the dominant tree is a late-successional feature, low dynamics characterizes the habitat.

Conclusions

The habitat type functions as meso-component level (category 2b) both in the CON and MED region.

Step 1.3 - Historical perspective: what happened to the habitat?

The markedly colder and drier climatic conditions of the Last Glacial Maximum must have caused the extinction of laurel populations located north of the Mediterranean Basin. Only a few regions functioned as main glacial refugia for the species from which it is currently spreading again: the Macaronesian Islands, the Iberian Peninsula, North Africa, southern Italy, the Aegean region, Transcaucasia and the Near East.

The particular ecological ‘micro-forest’ requirements of the habitat ensured its survival during the historical forest decline between 1000 and 1850. This is particularly true for those communities situated in remote sites, such as gorges and ravines. Stands of *L. nobilis* not protected by these terrain features were subjected much more to anthropogenic pressures. The traditional land use of forests such as wood pasture negatively affected the natural dynamics of the evergreen forest including laurel stands. However, during the past decades, the increase of forest cover along the Italian peninsula should favour the establishment of *L. nobilis* populations in new localities. Forestry measures can negatively affect this habitat, especially if the continuous canopy cover is interrupted. In central Italy the habitat cover area experienced a reduction of circa 10% due to forest management activities.

Step 1.4 - Analysis of distribution and trends

The habitat 5230* in Italy is currently mainly distributed along the Tyrrhenian coast and islands, and with scattered stands along the Adriatic coast. The current range has been estimated as 13,700 km^2 in biogeographical region MED and 300 km^2 in CON (Figure 9.1.1).

The European Red List of Habitats reported for the G2.2 Mainland laurophyllous woodland an overall increase of 30%, for 3 countries (Spain, France and Italy) over the last 50 years. The total area has passed from of 139 to 180 km^2 in this 50 year period. The biggest increase (+32%) has occurred in Portugal, that hosts 72% of the surface of the habitat [10]. The current and future trend is supposed to be stable. Historical trends (from 1750) remain unknown.

For Italy, G2.2 is similar to habitat type 5230*. Historical data are fragmented and an overall estimate is still missing for the distribution and trend of this habitat type. Based on expert opinion (see

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[10] https://forum.eionet.europa.eu/european-red-list-habitats/library/terrestrial-habitats/g.-forests/g2.2-mainland-laurophyllous-woodland

step 1.1, 1.2, 1.3 above) a negative trend in Italy due to forest exploitation and wood pasture for this habitat type is reported but naturally recovering patches have been spreading in recent times.

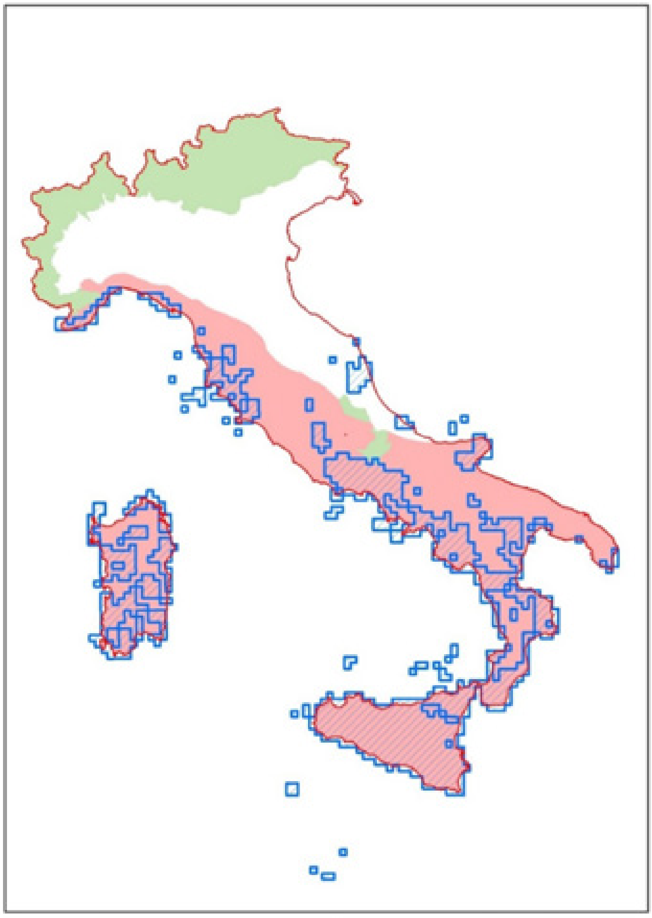

Figure 9.1.1 *Current distribution (in blue grids of 10x10 km squares) of the habitat type 5230* in Italy as reported in 2013 (biogeographical regions: green ALP; white CON and red MED)*

Conclusions

The area of habitat 5230* has declined in Italy by circa 10% in the last 50 years. This reduction is not due to changes in ecological requirements but the result of human pressures, mainly forest exploitation and wood-pasture. Although the absolute value of area loss is small with respect to the decrease of other forest habitats, the relative values are influential in terms of area occupancy. During the last decades, the recovery of forests in general has favoured the establishment of new patches of the habitat 5230* but its extent cannot be estimated yet.

Step 2.1 - FRA assessment

The habitat 5230* is decreasing but apparently recolonizing its natural range, due to land use changes. The area for Italy is 6.69 km^2 and was reported in 2013 as inadequate in MED (6.54 km^2) and favourable in CON (0.15 km^2). The potential area for the habitat, obtained by spatial modelling (see Supplement) is 1,821 km^2, only in MED. The distance to reference value is 1,814 km^2.

Reported values are based on expert opinion and highly underestimate (potential) area, probably only based on the contribution of the Natura2000 network (6.03 km^2).

Conclusions

FRA >> 6.69 km^2 (MED + CON).

Step 2.2 - FRR assessment

The range reported in 2013 as 13,700 km^2 in MED is considered Unfavourable - Inadequate (U1) and 300 km^2 in CON as Favourable. As previously noted, based on the ecological characteristics and variability of the habitat this value represents only 25% of the natural potential range.

Conclusions

FRR >> 14,000 km^2 (MED + CON).

Acknowledgements

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References

Agrillo, E., Alessi, N., Massimi, M., Spada, F., De Sanctis, M., Francesconi, F., Cambria, V.E. & Attorre F. 2016. Nationwide Vegetation-Plots Database – Sapienza University of Rome: State of the art, basic figures and future perspectives. Phytocoenologia DOI: http://dx.doi.org/10.1127/phyto/2016/0139.

Alessi A, Wellstein C, Spada F & Zerbe S, (2018) Phytocoenological approach to the ecology of Laurus nobilis L. in Italy. Rend. Fis. Acc. Lincei. https://doi.org/10.1007/s12210-018-0677-8.

Angelini, P., Casella, L., Grignetti, A., & Genovesi, P. (2016). Manuali per il monitoraggio di specifiee habitat di interesse comunitario (Direttiva 92/43/CEE) in Italia: habitat. ISPRA, Serie Manuali e linee guida, 142(2016), 280. (http://www.isprambiente.gov.it/it/pubblicazioni/manuali-e-linee-guida/manuali-per-il-monitoraggio-di-specie-e-habitat-di-interesse-comunitario-direttiva-92-43-cee-in-italia-habitat).

Bacchetta G., Bagella S., Biondi E., Farris E., Filigheddu R., Mossa L., 2004. A contribution to the knowledge of the order Quercetalia ilicis Br.-Bl. ex Molinier 1934 of Sardinia. Fitosociologia 41 (1): 29-51.

Bacchetta, G., Farris, E., Fenu, G., Filigheddu, R., Mattana, E., & Mulè, P. (2007). Contributo alla conoscenza dei boschi a Laurus nobilis L. della Sardegna, habitat prioritario ai sensi della Direttiva 92/43/CEE. Fitosociologia 44 (2): 239-244.

Burrascano S, Casavecchia S, Copiz R, Del Vico E, Galdenzi D, Gigante D, Lasen C, Spampinato G, Venanzoni R, Zivkovic L (2009a) Italian interpretation Manual of the habitats (92/43/EEC Directive). Ministero dell'Ambiente e della Tutela del Territorio e del Mare. http://vnr.unipg.it/habitat/.

Casella L, Bianco P.M., Angelini P & Morroni E., 2012. Italian National Vegetation Database (BVN/ISPRA). In: Vegetation databases for the 21st century. Biodiversity & Ecology, 4. ISSN: 1613-9801.

Chytrý, M., Hennekens, S. M., Jiménez-Alfaro, B., Knollová, I., Dengler, J., Jansen, F., ... & Ambarlı, D. (2016). European Vegetation Archive (EVA): an integrated database of European vegetation plots. Applied vegetation science, 19(1), 173-180.

EEA - Conservation status of habitat types and species ("Italian Habitats Directive Reporting" - Article 17, Habitats Directive 92/43/EEC) - https://www.eea.europa.eu/data-and-maps/data/article-17-database-habitats-directive-92-43-eec-1.

Falcucci, A., Maiorano, L., & Boitani, L. (2007). Changes in land-use/land-cover patterns in Italy and their implications for biodiversity conservation. Landscape Ecology 22: 617–631.

Filibeck, G. (2013). Le comunità a dominanza di Laurus nobilis in Italia: problemi interpretativi degli Habitat 5230* e 5310 della Direttiva 92/43/CEE. Archivio Geobotanico 14: 37-42.

Giacobbe, A. (1939). Ricerche geografiche ed ecologiche sul Laurus nobilis L. Archivio Botanico 15 (1): 33-82.

ISPRA, Corine land cover 2006 IV livello. Dati della Rete del sistema Informativo Nazionale Ambientale – SINAnet.

Lucchese, F., Pignatti, S. (1990). Sguardo sulla vegetazione del Lazio Marittimo. Quaderni dell'Accademia Nazionale dei Lincei 264: 5-48

Supplement: Modelling of the potential range
To estimate the potential of recovery of the natural range of habitat type 5230*, spatial modelling has been carried out, through the following steps:

- Selection of habitat type plots (see Figure 9.1.2A). The vegetation plots with characteristic/diagnostic species (i.e. by comparing the national and European habitat interpretation manual) were selected in the available national georeferenced vegetation databases (e.g. BVN-ISPRA and VDP-SAPIENZA). A further selection was made taking into account the structure of the habitat (relative cover of the diagnostic/characteristic species (e.g. equal or higher than 50% with respect to other tree species cover).
- Spatial modelling. Spatial predictive modelling was performed by Maxent, to estimate the geographic natural range of the habitat and the correlation with major environmental factors (e.g. seasonality and annual mean temperature, seasonality and annual precipitation, terrain elevation, slope and aspect).
- The preliminary result is a suitability map for the habitat 5230* across the Italian territory (i.e. "wider potential range"). The lower suitability values were excluded by using a threshold value provided by Maxent. Furthermore, territories occupied by patches of land that are not compatible with the existence of a natural habitat (i.e. urban areas, intensive and extensive agricultural area etc.) or patches not compatible with the habitat under examination (i.e. dunes, rivers etc.) were excluded.

The final result is a "potential range" map of habitat 5230* (see Figure 9.1.2B) of 52,600 km^2 (expressed as 10x10 km squares).

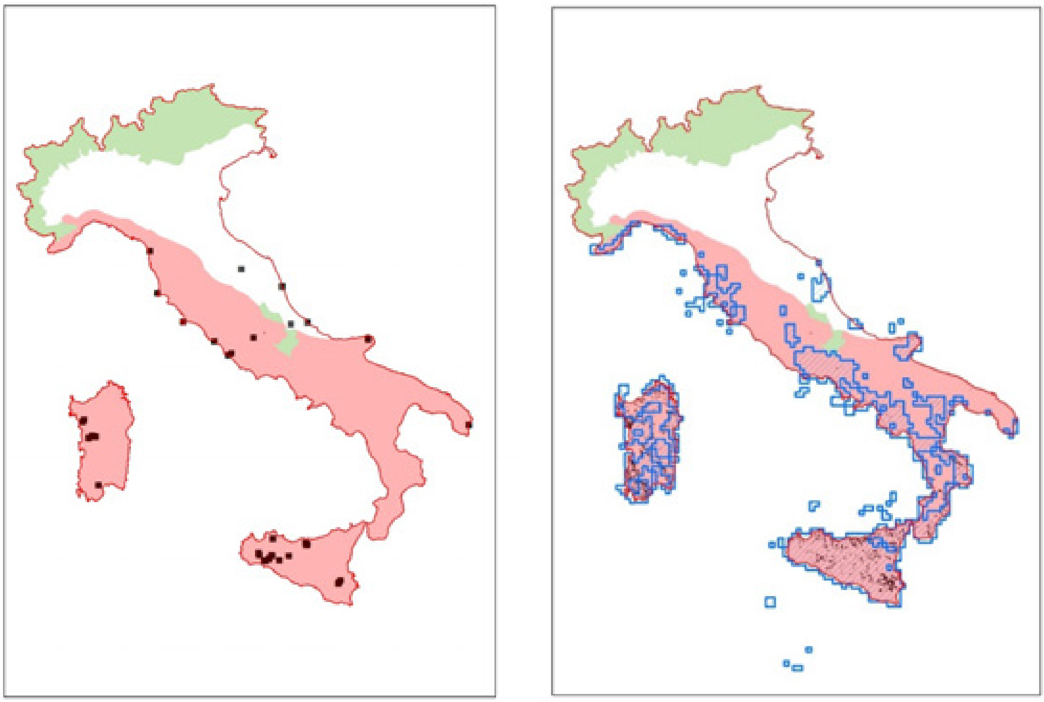

Figure 9.1.2 *Modelling of the potential range of 5230* in Italy. A (left). Selected plots for habitat 5230* on the map of the biogeographical regions (ALP green, CON white, MED red). B (right). Resulting potential range of habitat 5230* in black with current distribution outlined in blue*

9.2 Semi-natural dry grasslands and scrubland facies on calcareous substrates (*Festuco-Brometalia*) (6210*) in Italy

Emiliano Agrillo & Fabio Attorre

Step 1.1 - Ecology of the habitat
In Italy, the habitat type 6210* is represented by perennial grasslands dominated by hemi-cryptophytic grasses. These grasslands occur at low to medium altitudes. They are arid or semi-dry grasslands, sometimes with a high richness in orchids. In the latter case they are considered as priority habitat. The habitat is distributed mainly in the MED Region with 2/3 of of the total area; the remaining area is equally distributed in the CON and ALP regions.

At present in Italy a formal list of typical species for this habitat is not available. Dominant species are *Bromus erectus* and *Brachypodium pinnatum*. The species composition differs between the three biogeographical regions and is highly differentiated within each region as well. In MED diagnostic species are B*rachypodium genuense, Dactylorhiza sambucina, Anthyllis vulneraria, Anacamptis pyramidalis, Eryngium amethystinum, Festuca circumediterranea, Globularia meridionalis, Tragopogon crocifolius, Sideritis italica, Hippocrepis comosa, Ophrys spp., Orchis spp*. In CON, *Achillea tenorii, Allium spp., Asperula purpurea, Crepis lacera, Carex humilis, Centaurea ambigua, Erysimum pseudorhaeticum, Helianthemum apenninum*. In ALP, *Brachypodium phoenicoides, Arabis hirsuta, Bupleurum falcatum subsp. cernuum, Campanula glomerata, Centaurea nigra, Centaurea scabiosa, Knautia purpurea, Koeleria pyramidata, Leontodon hispidus, Salvia pratensis, Scabiosa columbaria*. In the Apennines the habitat can be rich in endemic species, and sometimes also in chamaephytes (dwarf shrubs). The habitat 6210* is important for butterflies, including *Melanargia arge, Eriogaster catax, Euplagia quadripunctaria, Euphydryas aurinia,* and *Zerynthia sp.*

Step 1.2 - Spatial scale of functioning
The habitat type is partly a natural grassland (e.g. high in the mountains, where many endemics are found), but mainly it consists of semi-natural grassland managed by mowing or grazing. Overgrazing causes the invasion by ruderal and nitrophilous species resulting in degraded habitat. The habitat shows a broad environmental range in the three biogeographical regions from arid to semi-mesophile grasslands. Consequently, the species pool varies considerably with different sets of continental or sub-mediterranean species. A great amount of papers have been produced to describe grassland communities belonging to the habitat 6210* (shown in the "Prodrome of the Italian vegetation" and addenda, where all the past revisions have converged, http://www.prodromo-vegetazione-italia.org/).

Conclusions

The habitat type functions at the meso-habitat level (category 2a).

Step 1.3 - Historical perspective: what happened to the habitat?
In Italy, these grasslands became more widespread during the glacial periods because of the more continental climate, but now they are mainly of secondary origin being promoted by the past agro-pastoral use of the landscape, particularly in the Apennines. In the last decades, due to the abandonment of traditional land use, they have suffered a significant reduction in area. Since this habitat is strictly dependent on extensive grazing, usually with sheep, and/or annual mowing, the abandonment started a succession to shrublands and woodlands. Moreover, grasslands on deep soils have been used for more intensive farming, with fertilising and reseeding, which leads to the disappearance of even more sites.

Abandonment and overgrazing are the main threats to this habitat. In particular during the second half of the 19th century, land use in European rural areas changed considerably, and grasslands were affected the most. The cessation of traditional management, due to both rural abandonment and agricultural intensification, resulted in the loss and fragmentation of habitats, which is considered one of the main causes of the European decrease in biodiversity. The management practices that affect biodiversity in grasslands are above all livestock grazing and mowing. The driving factors of these practices are the intensity of the activities, their timing, their seasonality and local environmental conditions. Another important but poorly-known factor is the kind and reuqired density of grazing animals (e.g. sheep, goats, cattle, horses). In order to preserve and restore the habitat, the following activities are suggested: annual mowing, clearing of smaller plantations on dry grasslands and establishment or continuation of appropriate grazing regimes.

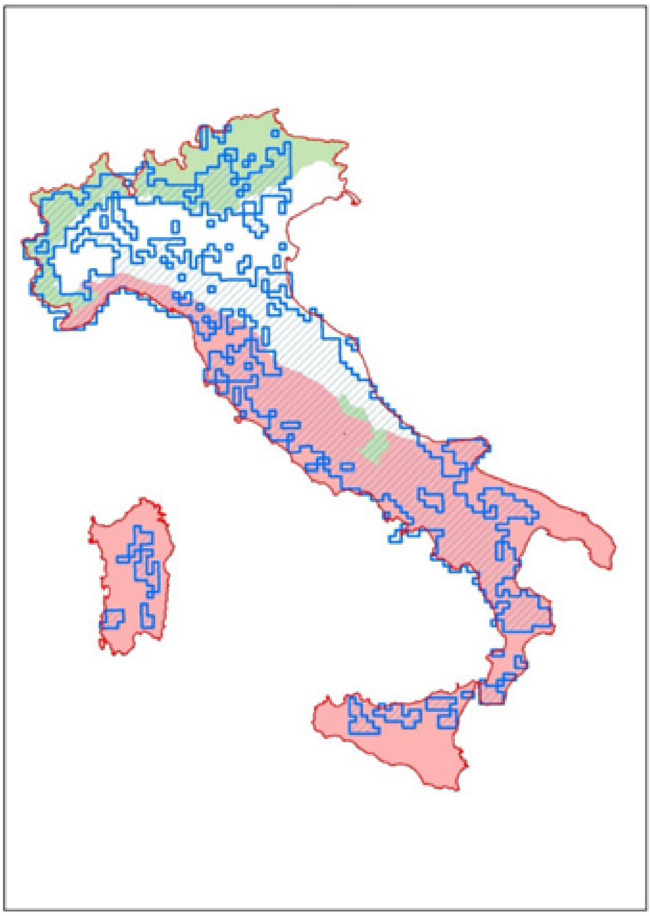

Figure 9.2.1 *Current distribution (in blue grids of10x10 km squares) of the habitat type 6210* in Italy as reported in 2013 (biogeographical regions: green ALP; white CON and red MED)*

Step 1.4 - Analysis of distribution and trends

Habitat type 6210* in Italy is distributed all over the peninsula, in all three biogeographical regions, but it is particularly spread in the Apennines. The actual total area reported in 2013 is 6406 km^2 estimated based on partial data with some extrapolation and/or modelling (see Figure 9.2.1): 4015,25 km^2 in MED Region, 1218,47 km^2 in CON and 1173,21 km^2 in ALP.

The Red List of European Habitats reports a short term negative trend of 40% in quantity of E1.2a (Semi-dry perennial calcareous grassland) for EU28[11]. Data on recent changes (since 1980), from local case studies (Northern and Central Apennine) reveal a 7 to 20% decrease; data on local historical changes (1954) in the Northern Apennine indicate a 50% decrease. Habitat type 6210* as part of E1.2a shares the decreasing trend in quantity and quality. Many Life projects have been implemented in the last decades in order to restore the habitat across Europe[12].

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[11] https://forum.eionet.europa.eu/european-red-list-habitats/library/terrestrial-habitats/e.-grasslands/e1.2a-semi-dry-perennial-calcareous-grassland-1

[12] see the European database of Life projects http://ec.europa.eu/environment/life/project/Projects/

By definition (see step 1.1), in Italy the habitat 6210* mainly occurs on limestone, but it can also colonize areas where other soils are intermixed with limestone. It represents one of the most widespread habitats of Community Interest and the surface area in Italy corresponds to 43% of the total European area for that habitat, even though it is in decline. In Italy the habitat shows a decline in distribution and in structure and functions, due to changes in land use. A significant area has been converted to woodlands by natural encroachment after agricultural abandonment in mountain areas. About 15% of the habitat area was lost since the 1950s and more dramatic percentages are known from local studies (see above).

Conclusions

In Italy the main threat to the grasslands of habitat 6210* is the abandonment of the traditional agro-pastoral activities, which in the last decades, in particular after the Second World War, resulted in a decline of about 15% of the open landscape and a subsequent spread of the forests, especially in mountain areas.

Step 2.1 - FRA assessment

The habitat 6210* is declining in area due to land use changes. In 2013 the area (6,406 km^2), was reported as inadequate with a decreasing trend in CON (1,218 km^2) and ALP (1,173 km^2), and a stable area in MED (4,015 km^2). The RefValue obtained by spatial modelling (see Supplement) is quite large, with a total RV1 (the distance to reference value) of about 10,000 km^2. Given the qualitative criteria applied for estimation only an operator can be used for the FRA assessment.

Conclusions

Overall: FRA >> 6,406 km^2.

Step 2.2 - FRR assessment

The current range is large enough to contain the required FRA and therefore it is considered as the FRR. The range reported in 2013 is 226,300 km^2 for the three biogeographical regions together (124,500 MED, 60,900 CON, 40,900 ALP), with an evaluation of "short-term trend decreasing" in CON and ALP and "stable" in MED. Total current values has an uncertainty between 147,800 and 226,300 km^2 (see Supplement). Values derived from the model are: MED=77600 km^2, CON=42200 km^2, ALP= 47900 km^2. So in the ALP region the modelled distribution is comparable with the reported value, while a great difference is reported for MED values. These differences can be due to expert judgement upon which the reporting was based, in particular for the MED region.

Conclusions

Overall: FRR is current value.

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We would like to thank Dr. Emanuela Carli for the technical support and Dr. Laura Casella (Italian National Institute for Environmental Protection and Research - ISPRA) for the careful reading and useful suggestions.

References

Abbate, G., Cesca, G., Codogno, M., Corbetta, F., & Puntillo, D. (1984). Sindinamica di fitocenosi prative della Sila Grande (Calabria). Biogeographia, 10, 161–178.

Agrillo, E., Alessi, N., Massimi, M., Spada, F., De Sanctis, M., Francesconi, F., Cambria, V.E. & Attorre F. 2016. Nationwide Vegetation-Plots Database – Sapienza University of Rome: State of the art, basic figures and future perspectives. Phytocoenologia DOI: http://dx.doi.org/10.1127/phyto/2016/0139.

Allegrezza, M. (2003). Vegetazione e paesaggio vegetale della dorsale del Monte San Vicino (Appennino centrale). Fitosociologia, 40(1 s1), 3–118.

Angelini, P., Casella, L., Grignetti, A., & Genovesi, P. (2016). Manuali per il monitoraggio di specifiee habitat di interesse comunitario (Direttiva 92/43/CEE) in Italia: habitat. ISPRA, Serie Manuali e linee guida, 142(2016), 280. (http://www.isprambiente.gov.it/it/pubblicazioni/manuali-e-linee-guida/manuali-per-il-monitoraggio-di-specie-e-habitat-di-interesse-comunitario-direttiva-92-43-cee-in-italia-habitat).

Bakker, E. S., Ritchie, M. E., Olff, H., Milchunas, D. G., & Knops, J. M. H. (2006). Herbivore impact on grassland plant diversity depends on habitat productivity and herbivore size. Ecology Letters, 9(7), 780–788.

Biondi, E., Allegrezza, M., Taffetani, F., Ballelli, S., & Zuccarello, V. (2002). Excursion to the National Park of Gran Sasso and Monti della Laga. Fitosociologia, 39 (1)(suppl. 3), 43–59.

Biondi, E., Allegrezza, M., & Zuccarello, V. (2005). Syntaxonomic revision of the Apennine grasslands belonging to Brometalia erecti, and an analysis of their relationships with the xerophilous vegetation of Rosmarinetea officinalis (Italy). Phytocoenologia, 35, 129–164.

Biondi, E., Ballelli, S., Allegrezza, M., Guitian, J., & Taffetani, F. (1986). Centaureo bracteatae-Brometum erecti ass. nova dei settori marnoso-arenacei dell'Appennino centrale –. Documents Phytosociologiques. Phytosociol, 10, 117–126.

Biondi, E., Ballelli, S., Allegrezza, M., & Zuccarello, V. (1995). La vegetazione dell'ordine Brometalia erecti Br.-Bl. 1936 nell'Appennino (Italia). Fitosociologia, 30, 3–45.

Biondi, E., Ballelli, S., & Principi, D. (1985). Sur les pelousesseches des substratsmarneux- arenaces de l'Apenninseptentrional (Italie). Documents Phytosociologiques N.S., 9, 351–357.

Biondi, E., Blasi, C., Allegrezza, M., Anzellotti, I., Azzella, Martin, M., Carli, E., … Zivkovic, L. (2014). Plant communities of Italy: The Vegetation Prodrome. Plant Biosystems - An International Journal Dealing with All Aspects of Plant Biology, 148(4), 728–814.

Biondi, E., & Galdenzi, D. (2012). Phytosociological analysis of the grasslands of Montagna dei Fiori (central Italy) and syntaxonomic review of the class Festuco-Brometea in the Apennines. Plant Sociology, 49(1), 91–112.

Biondi E., B. S. (1982). La vegetation du Massif du Catria (Apennin central) avec carte phytosociologique 1: 15.000. In F. Pedrotti (Ed.), Guide-Itineraire - Excursion Internationale de Phytosociologie en Italie centrale (pp. 211–236).

Blasi, C., Facioni, L., Burrascano, S., Del Vico, E., Tilia, A., & Rosati, L. (2012). Submediterranean dry grasslands along the Tyrrhenian sector of central Italy: synecology, syndynamics and syntaxonomy. Plant Biosystems, 146, 266–290.

Brullo, S., & Grillo, M. (1978). Ricerche fitosociologiche sui pascoli dei Monti Nebrodi (Sicilia settentrionale). Notiziario Della Soietà Italiana Di Fitosociologia, 13, 23–61.

Burrascano S, Casavecchia S, Copiz R, Del Vico E, Galdenzi D, Gigante D, Lasen C, Spampinato G, Venanzoni R, Zivkovic L (2009a) Italian interpretation Manual of the habitats (92/43/EEC Directive). Ministero dell'Ambiente e della Tutela del Territorio e del Mare. Http://vnr.unipg.it/habitat/.

Calaciura, B., & Spinelli, O. (2008). Management of Natura 2000 habitats. 6210 Semi-natural dry grasslands andscrubland facies on calcareous substrates (Festuco-Brometalia) (*important orchid sites). European Commission. Technical Report 2008 12/24.

Casella L, Bianco P.M., Angelini P & Morroni E., 2012. Italian National Vegetation Database (BVN/ISPRA). In: Vegetation databases for the 21st century. Biodiversity & Ecology, 4. ISSN: 1613-9801.

Chytrý, M., Hennekens, S. M., Jiménez‐Alfaro, B., Knollová, I., Dengler, J., Jansen, F., ... & Ambarlı, D. (2016). European Vegetation Archive (EVA): an integrated database of European vegetation plots. Applied vegetation science, 19(1), 173-180.

de Bello, F., Lepš, J., & Sebastià, M.-T. (2007). Grazing effects on the species-area relationship: Variation along a climatic gradient in NE Spain. Journal of Vegetation Science, 18(1), 25–34.

Di Pietro, R. (2011). New dry grassland associations from Ausoni-Aurunci mountains (central Italy). Syntaxonomical updating and discussion on the higher rank syntaxa —. Hacquetia, 10(2), 183–231.

Díaz, S., Lavorel, S., Mcintyre, S., Falczuk, V., Casanoves, F., Milchunas, D. G., Campbell, B. D. (2007). Plant trait responses to grazing ? a global synthesis. Global Change Biology, 13(2), 313–341.

EEA - Conservation status of habitat types and species ("Italian Habitats Directive Reporting" - Article 17, Habitats Directive 92/43/EEC) - https://www.eea.europa.eu/data-and-maps/data/article-17-database-habitats-directive-92-43-eec-1.

EEA. (2015). State of nature in the EU Results from reporting under the nature directives 2007–2012. EEA. Retrieved from http://www.eea.europa.eu/publications/state-of-nature-in-the-eu

Falcucci, A., Maiorano, L., & Boitani, L. (2007). Changes in land-use/land-cover patterns in Italy and their implications for biodiversity conservation. Landscape Ecology, 22(4), 617–631.

Foggi, B., Lastrucci, L., Gennai, M., & Viciani, D. (2014). The Festuco-Brometea grasslands on sandstone and marl-clay-sandstone substrata in Tuscany (northerncentral Italy) —. Hacquetia2, 13(1), 19–54.

Galvánek, D., & Lepš, J. (2009). How do management and restoration needs of mountain grasslands depend on moisture regime? Experimental study from north-western Slovakia (Western Carpathians). Applied Vegetation Science, 12(3), 273–282.

Gibson, D. J. (2009). Grasses and Grassland Ecology. Oxford University Press.

Hobohm, C. (2014). Endemism in Vascular Plants. Plant and Vegetation (Vol. 9).

ISPRA, Corine land cover 2006 IV livello. Dati della Rete del sistema Informativo Nazionale Ambientale – SINAnet.

Klimek, S., Richtergenkemmermann, A., Hofmann, M., & Isselstein, J. (2007). Plant species richness and composition in managed grasslands: The relative importance of field management and environmental factors. Biological Conservation, 134(4), 559–570.

Lucchese, F., Persia, G., & Pignatti, S. (1995). I prati a Bromus erectus Hudson dell'Appennino Laziale. Fitosociologia1, 30, 145–180.

Moreira, F., & Russo, D. (2007). Modelling the impact of agricultural abandonment and wildfires on vertebrate diversity in Mediterranean Europe. Landscape Ecology, 22(10), 1461–1476.

Pärtel, M., Bruun, H. H., & Sammul, M. (2005). Biodiversity in temperate European grasslands: origin and conservation. In Grassland Science in Europe (pp. 1–14). Tartu (Estonia): Greif printhouse.

Pietro, R. Di, Theurillat, J., Capelo, J., Fernández-, F., Terzi, M., Andra, C., … Nomenclature, L. (2015). Nomenclature and syntaxonomic notes on some high-rank syntaxa of the European grassland vegetation. Lazaroa, 36, 79–106.

Poschlod, P., & WallisDeVries, M. F. (2002). The historical and socioeconomic perspective of calcareous grasslands—lessons from the distant and recent past. Biological Conservation, 104(3), 361–376.

Pykälä, J., Luoto, M., Heikkinen, R. K., & Kontula, T. (2005). Plant species richness and persistence of rare plants in abandoned semi-natural grasslands in northern Europe. Basic and Applied Ecology, 6(1), 25–33.

WallisDeVries, M. F., Poschlod, P., & Willems, J. H. (2002). Challenges for the conservation of calcareous grasslands in northwestern Europe: integrating the requirements of flora and fauna. Biological Conservation, 104(3), 265–273.

Wilson, J. B., Peet, R. K., Dengler, J., & Pärtel, M. (2012). Plant species richness: The world records. Journal of Vegetation Science, 23(4), 796–802.

Supplement: Modelling of the potential range

To estimate the potential of recovery of the natural range of habitat type 6210*, spatial modelling has been carried out, through the following steps:

- Selection of habitat type plots (see Figure 9.2.2A). The vegetation plots with characteristic/diagnostic species (i.e. by comparing the national and European habitat interpretation manual) were selected in the available national georeferenced vegetation databases (e.g. BVN-ISPRA and VDP-SAPIENZA). A further selection was made taking into account the structure of the habitat (relative cover of the diagnostic/characteristic species (e.g. equal or higher than 50% with respect to other tree species cover).
- Spatial modelling. Spatial predictive modelling was performed by Maxent, to estimate the geographic natural range of the habitat and the correlation with major environmental factors (e.g. seasonality and annual mean temperature, seasonality and annual precipitation, terrain elevation, slope and aspect).
- The preliminary result is a suitability map for the habitat 6210* across the Italian territory (i.e. "wider potential range"). The lower suitability values were excluded by using a threshold value provided by Maxent. Furthermore, territories occupied by patches of land that are not compatible with the existence of a natural habitat (i.e. urban areas, intensive and extensive agricultural area etc.) or patches not compatible with the habitat under examination (i.e. dunes, rivers etc.) were excluded.

The final result is a “potential range” map of habitat 6210* (see Figure 9.2.2B) of 147,800 km^2 (expressed as 10x10 km squares).

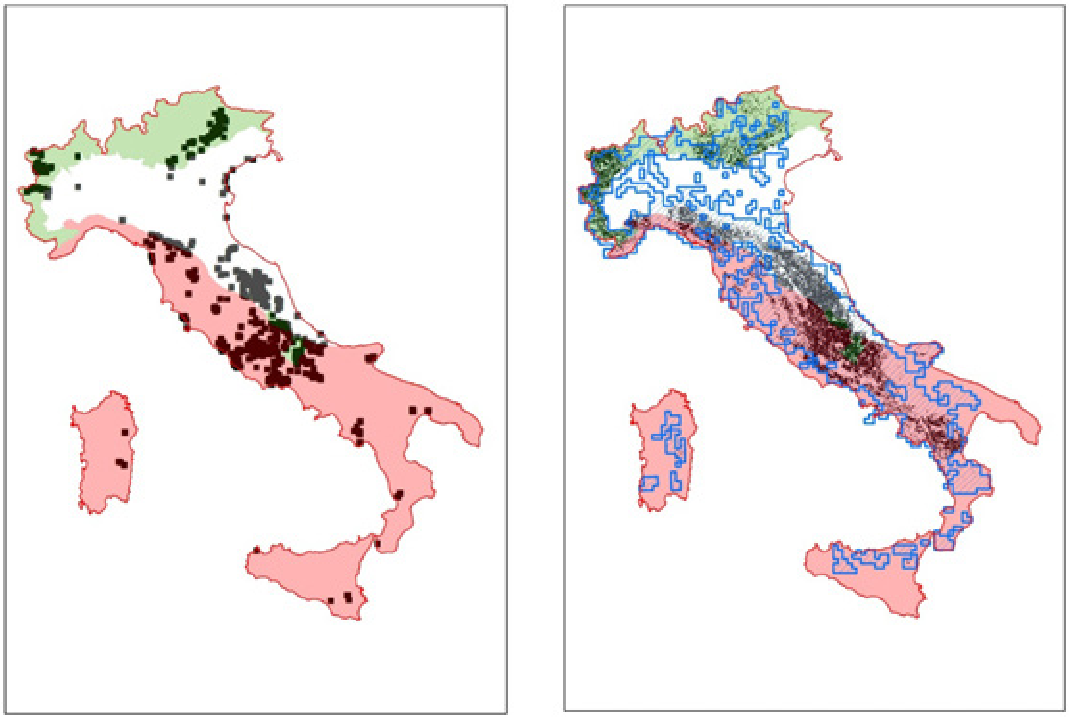

Figure 9.2.2 *Modelling of the potential range of 6210* in Italy. A (left). Selected plots for habitat 5230* on the map of the biogeographical regions (ALP green, CON white, MED red). B (right). Resulting potential range of habitat 6210* in black with current distribution outlined in blue*

9.3 Molinia meadows on calcareous, peaty or clayey-silt-laden soils (*Molinion caeruleae*) (6410) in the Netherlands

John Janssen & Rienk-Jan Bijlsma

Step 1.1 - Ecology of the habitat
Habitat type 6410 in the Netherlands consists of mesotrophic fen meadows (alliance *Junco-Molinion*, mainly association *Cirsio-Molinietum*) characteristic of moist to wet and slightly acid to neutral sites with a groundwater table at or sligtly above the soil surface during winter. These meadows are very low productive due to a low nitrogen and phosphorous availability (Jansen 2000). Typical species include the butterfly Boloria selene and several vascular plants species e.g. *Carex hostiana*, *C. pulicaris* and *Viola persicifolia*.

Step 1.2 - Spatial scale of functioning
The habitat consists of semi-natural grassland with a relatively homogeneous vegetation structure. Annual mowing (hay making) is needed for maintaining the structure and species composition and preventing succession towards shrubland and forest. The habitat shows ecological variation in different landscape types (Jansen 2000). It occurs in landscapes on sandy soils influenced by relatively base-rich groundwater or by flooding with buffered surface water (on peaty soils). The habitat functions optimally in gradients with other grassland types or dwarf shrubland, like *Calthion* hay meadows or Alkaline fens (H7230) on the lower, wetter sides and *Nardus* grasslands (H6230) and heathlands on the higher, drier side (H4030) or in more acidic, moist conditions (H4010). In such a landscape configuration many of the characteristic species built up large populations over different types of grasslands or habitats, while they can fluctuate between different habitats if some of the conditions (like water table) change.

Conclusions

The habitat type functions at the meso-habitat level (category 2a).

Step 1.3 - Historical perspective: what happened to the habitat?
In the 19th century in the Netherlands the main plant community of the habitat (association *Cirsio-Molinietum*) covered several 1000's of square km (Sissingh 1978). About 1920 some 100's of km^2 remained, but afterwards many of these sites have disappearde due to land reclamation, increased fertiliser application and changes in water management. Based on the relative amount of relevés in a country-wide grassland survey between 1940 and 1970 (De Vries *et al.* 1942, 1953; Kruijne *et al.* 1967) it is estimated that around WW II about 1690 ha of these grasslands remained (Weeda 2014). The destruction of this habitat continued until about 1975 (Weeda *et al.* 2002). The area of the plant community *Cirsio-Molinietum* at that moment was about 135 ha.

The total area of the habitat is larger, as the type is a somewhat broader defined, including some *Juncus acutiflorus* communities. About these communities little historical information is available, but it is likely that they covered much smaller areas than the *Cirsio-Molinietum* (Weeda 2014). At present the total area of the habitat in the Netherlands is assessed to be 286 ha (Weeda 2014).

Nowadays, many H6410 sites are rather isolated and small, and the habitat is considered to occur very fragmented (Figure 1). This is reflected in the fauna. Already in the middle of the 20th century the species composition of the habitat had declined largely, especially in terms of entomofauna (Weeda 2014). This is illustrated by the extinction of four insect species in the Netherlands which live on *Succisa pratensis* and were largely restricted to this habitat: *Euphydryas aurinia*, *Hemaris tityus*, *Andrena marginata* en *Nomada argentata*. The first three were still rather widespread at the beginning of the 20th century (Meerman 1987, Bos *et al.* 2006, Peeters *et al.* 2016). Another characteristic species, the butterfly *Boloria selene*, is endangered in the Netherlands. For sustainable populations of characteristic fauna species large complexes of the habitat are needed. The characteristic plant species *Gymnadenia conopsea* has (almost) disappeared from this habitat, while many others are threatened (*Carex pulicaris*, *Carex buxbaumii*, *Carex hostiana*, *Cirsium dissectum*, *Scutellaria minor*, *Carum verticillatum*, *Viola persicifolia*).

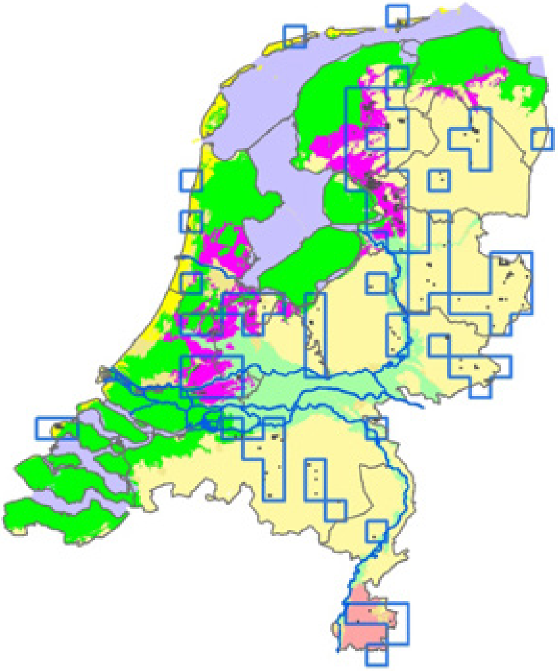

Figure 9.3.1 *Distribution (1 km squares as black dots) and range (corresponding 10x10 km squares in blue) of habitat type 6410 in the Netherlands (as reported in 2013)*

Step 1.4 - Analysis of distribution and trends
The negative trend since 1920 (over 100 year) is more than 95% and since 1940 (75 year) more than 75%, so larger than 1% per year (Table 9.3.1). More recent trends are less relevant, as the habitat was not functioning optimally anymore around 1950 (see above). The period before 1950 is choosen as reference for a good functioning habitat, i.e. 1690 ha.

Table 9.3.1 *Trends in area of habitat 6410 in the Netherlands*

	Area H6410 (km^2) *Cirsio-Molinietum*	 *Crepido-Juncetum* (oligotrophic part)	 Total estimate
19th cent.	>>1000	?	1500
1920	>>100	?	200
1956	1.35	1	2.4
2010	1.35	1	2.4

The overall range has only slightly changed during the last century, and the habitat is still found in all regions of its historical distribution except in western Noord-Brabant (Weeda 2014). On a more detailed scale than 10x10 grids, more gaps in the range have occurred (Figure 9.3.1).

Because of the fragmented character of the habitat and the poor condition of many characteristic flora and fauna species, there is a clear need of a larger area. Area increase should occur mainly in the direct surrounding of existing sites, focussing on restoring large complexes of this kind of grasslands that are suitable for the typical fauna.

Conclusions

This semi-natural grassland type declined strongly in range and area with a corresponding loss in species compisiton, especially envertebrates. The main causes are land reclamation, increased fertiliser application and changes in water management.

In the recent past, between 1900 and 1950, range and area showed strong negative trends (>1% per year).

Step 2.1 - FRA assessment

Little is known about functioning of the habitat in relation to its area. Therefore, we use historical data for assessing the FRA and apply a reference-based approach. The strong decline in area relative to the reference value 1690 ha requires a large part of the 'lost area' (1690 - 286 ha) to be restored, of which 25-75% seems ecologically feasible.

Conclusions

FRA = 6.4-13 km^2 (FRA >> CV) (Weeda 2014).

Step 2.2 - FRR assessment

Current range is 10,900 km^2. The range needs to be increased to restore the historical ecological variation, especially in the western part of the province Noord-Brabant, where the habitat has disappeared. An increase of about 4 grid cells, including that region, is needed to meet the FRR. This seems ecologically feasible.

Conclusions

FRR=11,300 km^2 (FRR > CV) (Weeda 2014).

References

Bos, F., M. Bosveld, D. Groenendijk, C. van Swaay, I. Wynhoff & De Vlinderstichting (2006). De dagvlinders van Nederland, verspreiding en bescherming (Lepidoptera: Hesperioidea, Papilionoidea). Nederlandse Fauna 7. Naturalis, KNNV Uitgeverij & EIS, Leiden.

De Vries, D.M., M.L. 't Hart & A.A. Kruijne (1942). Een waardering van grasland op grond van de plantkundige samenstelling. Landbouwkundig Tijdschrift 54: 245-265.

Jansen, A.J.M. (2000). Hydrology and restoration of wet heathlands and fen meadow communities. PhD Thesis, Rijksuniversiteit Groningen.

Kruijne, A.A., D.M. de Vries & H. Mooi (1967). Bijdrage tot de oecologie van de Nederlandse graslandplanten. Verslagen van Landbouwkundige Onderzoekingen 696. Mededeling 338 van het I.B.S. Centrum voor landbouwpublikaties en landbouwdocumentatie, Wageningen.

Meerman, J.C. (1987). De Nederlandse Pijlstaartvlinders (Lepidoptera: Sphingidae). Wetenschappelijke Mededeling KNNV 180.

Peeters, T.M.J., H. Nieuwenhuijsen, J. Smit, F. van der Meer, I.P. Raemakers, W.R.B. Heitmans, C. van Achterberg, M. Kwak, A.J. Loonstra, J. de Rond, M. Roos & M. Reemer (2012). De Nederlandse bijen (Hymenoptera: Apidae s.l.). Natuur van Nederland. Naturalis Biodiversity Centre & European Invertebrate Survey, Leiden.

Sissingh, G. (1978). Le Cirsio-Molinietum Sissingh et De Vries (1942) 1946 dans les Pays-Bas. In: J.M. Géhu (red.), Les prairies humides. Colloques phytosociologiques 8. Cramer, Vaduz, pp. 289-301.

Weeda, E.J. (2014). H6410 Blauwgraslanden. In: Bijlsma, R.J., J.A.M. Janssen, E.J. Weeda & J.H.J. Schaminée. Gunstige referentiewaarden voor oppervlakte en verspreidingsgebied van Natura 2000-habitattypen in Nederland. WOt-rapport 125, Wettelijke Onderzoekstaken Natuur & Milieu, Wageningen: 134-137.

9.4 Alkaline fens (7230) in the Netherlands

John Janssen

Step 1.1 - Ecology of the habitat
This fen habitat, dominated by small *Cyperaceae* and brown-mosses, occurs in base-rich seepage areas, often in an altitudinal gradient in the landscape, but also in sandy depressions with seepage of alkaline groundwater. For maintaining the structure and species composition and preventing succession towards shrub and forest, yearly hay making is needed, as groundwater discharge in the lowlands is usually relatively low (compared to montane regions).

The habitat is found in a range of different landscapes, which results in diversity in its species composition and functioning. In the hilly regions of Limburg and Twente the habitat occurs in seepage areas, in the eastern flank of the Veluwe glacial ridge and in broader river floodplains, the habitat is found in depressions with discharge of water. In the floodplains such (usually sandy) sites remain after the removal (or exploitation) of clay. In some eastern regions (Achterhoek, Twente) the habitat is found in groundwater fed depressions in heathlands, while the habitat is historically known from a site on Muschelkalk as well. One other site is the result of human activities: a site where calcareous water discharges from a canal (similar conditions are known from a Flemish site).

The habitat has a relatively long list of characteristic vascular plants and (brown) mosses. Many of these are threatened, and therefore on the national red lists. Some characteristic species are restricted to certain regions: *Carex dioica* to the northeast of the Netherlands, *Pinguicula vulgaris* to the east, *Carex flava* to the south.

Step 1.2 - Spatial scale of functioning
The habitat occurs on very specific, usually small spots, often in a gradient towards other fen habitats, wet heath and/or grasslands. It may be dependent on large and deep groundwater bodies, or exist in situations with smaller and more shallow groundwater systems. As long as the hydrological conditions (regular discharge of unpolluted and – in most cases – base-rich groundwater) are not altered and a suitable management is maintained, the habitat structure and function and site area remain rather stable. For this reason, though the area of functioning is very small, the habitat is considered to function at a meso-scale.

Conclusions

The habitat type functions at the meso-habitat level (category 2a).

Step 1.3 - Historical perspective: what happened to the habitat?
During the 20th century the habitat has disappeared from several regions in the Netherlands. In South-Limburg, one of the regions with a historical concentration of sites, the number of sites declined for about 90%. The habitat decline is illustrated by negative trends in (inland) distribution of most of the typical vascular plant species (Figure 9.4.1), for instance *Epipactis palustris*, *Dactylorhiza incarnata*, *Parnassia palustris*, *Pinguicula vulgaris*, *Carex dioica*, *Eriophorum latifolium* and *Carex flava* (Hilgers 1972; Willems 1982; Van der Meijden & Holverda 2007; Weeda 2008). Their decline ranges from 88 to 97% of the historical occurrences (Van der Meijden *et al.* 2000). At least one characteristic species got extinct (*Spiranthes aestivalis*). Also some of the characteristic moss species have become very rare.

In the remaining sites the area of the habitat usually is very small, ranging from several ares to about 3 ha maximum. Because of the flat character of the landscape in the Dutch lowlands (Jansen 2000), it is likely that historical occurrences usually covered relatively small areas as well. Because of this typical small size of the habitat, we use distribution of the habitat as a measure for decline in and restoration of quantity, rather than area. Preferably we use data on a fine spatial scale, like distribution in 1x1 km-grids.

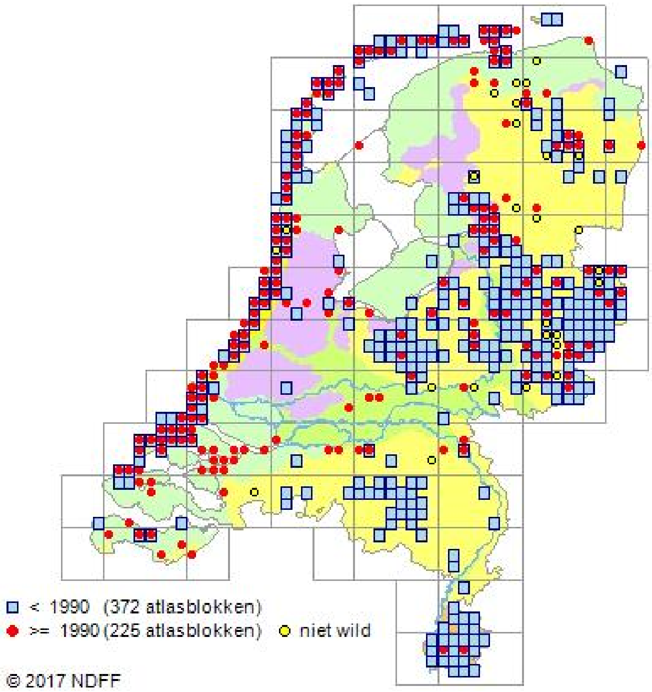

Figure 9.4.1 *Distribution of Parnassia palustris before and after 1990 in 5x5 km-grids. Inland occurrences in most cases indicate the presence of habitat 7230. The changes in inland species distribution illustrates that the habitat has strongly declined. The – more stable – coastal occurrences of the species indicate the presence of habitat type 2190 (dune slacks)*

Step 1.4 - Analysis of distribution and trends

The reference year is defined as the period 1930-1950. Around 1930 started the huge "agricultural improvements" in the Pleistocene sand landscapes of the Netherlands, though in some regions these land use changes started only in the early 1960s. Using the distribution of typical plant species as an indicator, the historical area in the period 1930-1950 is estimated to be at least 10 times the present value. The decline in distribution grids is more than 90% over the last 80 years, more than 1% per year. In the period 2003-2012 the habitat was reported from 21 grids of 10x10 km^2. Most of these relatively large grids relate only to one site. The total area was estimated to be only 0.11 km^2.

Successful restoration of the habitat is possible, as was shown by several projects in cover sand areas (Weeda *et al.* 2006) and along the rivers (Kerkhof 2016), as long as the hydrological conditions are suitable (alkaline seepage). Several characteristic species returned in restored or enlarged sites, both from the seed bank or from neighboring areas, even in cases where the hydrological systems are relatively small (Jansen 2000).

Conclusions

A large increase of FRA and FRV is required to reach a situation in which the habitat forms a sustainable network in which the characteristic plant species can live. Restoration of the habitat is feasible and well possible, as was shown in several projects.

Step 2.1 - FRA assessment

Because of the fragmented character of the habitat and the poor condition of most of its characteristic flora species, there is a clear need of a larger distribution and area for good functioning. As the decline has been extremely large, and most of the characteristic species have become threatened, a relatively large increase in area (and distribution) is required, to establish a network of sites in which the habitat can function sustainably. Following the assessment of Weeda (2014), we set the FRA at a value of present distribution plus 75-100% of the declined distribution. Using distribution as a proxy of favourable area, this means an increase of 15 to 21 sites. In terms of area it means an increase towards 0.19-0.22 km^2, but this should be realized over a network of sites.

Conclusions

FRA = 36-42 sites in 1x1 km-grids, or 0.19-0.22 km^2 realized over an extensive network of sites.

Step 2.2 - FRR assessment

It is likely that an increase of range (in terms of 10x10 km^2-grids) is required to realize the increase of area and distribution. It is roughly estimates that a range increase of 30% is needed to reach the favourable status. The increase should be spread more-or-less evenly over the whole range of the habitat.

Conclusions

FRR = 2700 km^2.

References

Bijlsma, R.J., J.A.M. Janssen, E.J. Weeda & J.H.J. Schaminée (2014). Gunstige referentiewaarden voor oppervlakte en verspreidingsgebied van Natura 2000-habitattypen in Nederland. WOt-rapport 125, Wageningen.

Hilgers, J.H.M. (1972). De achteruitgang van de Orchidaceae in Zuid-Limburg XII. Natuurhistorisch Maandblad 61: 54-56.

Jansen, A.J.M. (2000). Hydrology and restoration of wet heathlands and fen meadow communities. Dissertatie Rijksuniversiteit Groningen.

Kerkhof, D. (2016). Pioniergemeenschappen de Geeren en Lekuiterwaarden. PKN Excursieverslag (www.stratiotes.net).

Van der Meijden, R. & W.J. Holverda (2006). Revisie van het NHN-herbariummateriaal van Carex lepidocarpa Tausch (Schubzegge) en Carex flava L. (Gele zegge) in Nederland. Gorteria 31: 129-136.

Weeda, E.J. (2008). Plantensociologische positie van Cyperaceae en Juncaceae in hellingmoerassen in Zuid-Limburg. Stratiotes 36/37: 15-60.

Weeda, E.J., S.M.A. Keulen & J.W. Koelink (2006). Maaibeheer in de Kathager Beemden beloond: Veenzegge (Carex davalliana Sm.) nieuw voor Nederland. Natuurhistorisch Maandblad 95: 262-268.

Weeda, E.J., H. de Mars & S.M.A. Keulen (2011). Kalkmoeras in Zuid-Limburg. Natuurhistorisch Maandblad 100: 233-242.

Willems, J.H. (1982). Parnassia palustris L. in Zuid-Limburg. Gorteria 11: 99-106.

9.5 Atlantic acidophilous beech forests with Ilex and sometimes also Taxus in the shrublayer (*Quercinion robori-petraeae* or *Ilici-Fagenion*) (9120) in the Netherlands

Rienk-Jan Bijlsma & John Janssen

Step 1.1 - Ecology of the habitat
In the Netherlands habitat type 9120 is defined as being restricted to 'old forest sites' on dry, acidic soils, but slightly more base-rich (often loamy sands or loams) than the related habitat 9190 on very nutrient poor, sandy soils. 'Old forest sites' refer to native woodland dating from 1850 or older including adjacent more than 100 year old stands (Bijlsma *et al.* 2010).

The habitat is broadly defined in terms of dominant tree species, which besides beech (*Fagus sylvatica*) includes oak (*Quercus robur*, *Q. petraea*), birch (*Betula pendula*, *B. pubescens*) or sycamore (*Acer pseudoplatanus*). Beech is the most shade-tolerant and competitive species, spreading commonly in adjacent heathland and drift sand afforestations. The herb layer typically consists of bracken (*Pteridium*), brambles (*Rubus*), honeysuckle (*Lonicera periclymenum*), blue berry (*Vaccinium myrtillus*) and other acidophytic species. Holly (*Ilex*) is not considerd diagnostic because all forest types in the region are invaded now by this species (often as garden escape) which benefits from climate warming as well (Walther *et al.*, 2005). Typical species are the lichen *Lecanactis abietina*, confined to ancient beech woodland, the vascular plants *Convallaria majalis*, *Maianthemum bifolium*, *Oxalis acetosella* and *Polygonatum multiflorum*, and the fauna species slow worm (*Anguis fragilis*), nuthatch (*Sitta europaea*) and black woodpecker (*Dryocopus martius*).

Step 1.2 - Spatial scale of functioning
This forest type, like other temperate (semi-natural) ancient deciduous woodlands in western Europe, can show gap scale dynamics (~25 yr cycle), stand scale dynamics (~200 yr cycle) as well as long-term succession (millenium scale) (see e.g. Hahn *et al.*, 2007). It comprises developmental and successional stages such as gaps, rejuvenating sites with light-demanding species, old-growth with coarse woody debris and glades, edge habitats and gradients to grasslands and heathlands. This heterogeneity and dynamics is important for characteristic species and can only occur and develop in more or less continuous areas of woodland habitat from 10s to 100s of ha.

Conclusions

The habitat type functions at the macro-habitat level (category 1a).

Step 1.3 - Historical perspective: what happened to the habitat?
The rate of decline of forest cover in the Netherlands is quite probably similar as found for Denmark and the UK (Kaplan *et al.* 2009): from a low 20% about 1200 AD to even lower than 10% in the 18th century followed by slow recovery afterwards (to current 11%) due to heathland and drift sand afforestations, mostly with Scots pine, Douglas fir and larch. In the early 19th century woodlands on sandy and loamy soils, typical for 9120, occurred as islands, often near settlements, in vast areas of heathland. Most of these forests were oak-coppice used for firewood and more importantly for leather tanning (oak bark). Beech was intentionally suppressed in these coppices but remained present in scattered remnants of high forests on the better soils. Holly almost disappeared from coppiced woodlands in these open landscapes (during the so-called Little Ice Age) due to its low frost tolerance (Pott 1990).

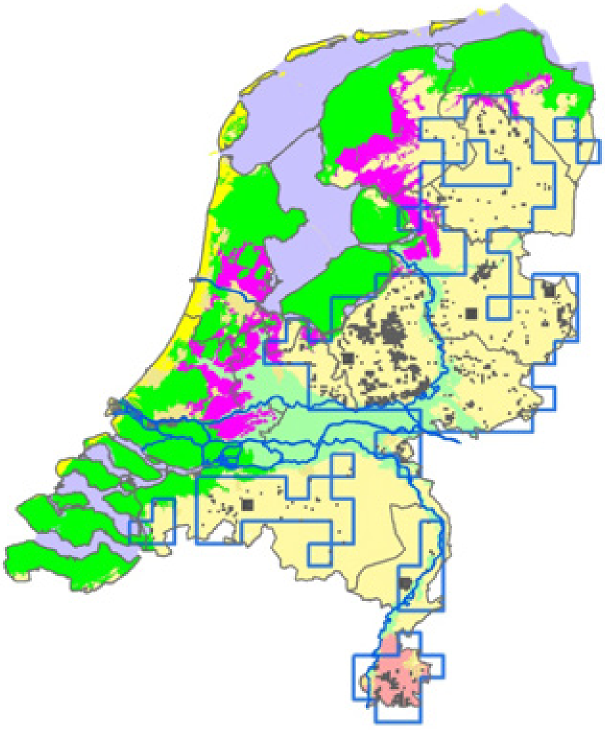

Figure 9.5.1 *Distribution (1 km squares as black dots) and range (corresponding 10x10 km squares) of habitat type 9120 in the Netherlands (as reported in 2013). The potential range is more or less confined to the physical-geographic region 'higher sandy soils' (pale brown)*

Between 1900 and 1940 oak coppice was mostly abandoned as management practice because of the availability of chemical alternatives for leather tanning and the replacement of firewood by fossil fuels. Coppices were transformed into high forest or converted to more productive coniferous (plantation) forest. At the sime time, large scale heathland afforestatons took place and continued to about 1960. Until then almost all forest was more or less intensively used. In the early 1980s, after much debate, nature-based principles for forest management were introduced. Since then natural (spontaneous) forest development and dynamics, dead wood and natural rejuvenation became accepted as part of more nature-based forest management. A network of 60 forest reserves was established in all potential natural forest types across the country.

By definition (see step 1.1), habitat 9120 in the Netherlands is more or less rectricted to the physical geographic region Higher sandy soils (~16350 km^2, about 50% of the land area; Figure 1) and within this region to well-drained soils in loamy sand or loam (~5400 km^2). The actual area has been estimated as 123 km^2 with 6750 ha within the Natura 2000 network. About 90% of this area is found in just one site, the Veluwe, in the centre of the country (Bijlsma 2014; and see Figure 9.5.1).

Step 1.4 - Analysis of distribution and trends

The historical perspective (step 1.3) implies a strong negative trend for area in the historical past (Middle Ages) and a corresponding drop in structure & function. Although the natural range is probably still rather intact, distribution and area within this range are highly fragmented with more or less continuous occurrences (at 1 km square-level) only in the central Veluwe-area (Figure 9.5.1).

As a consequence of the Dutch definition of 9120 (restricted to native, deciduous woodland older than 1850 and adjacent stands older than 100 yr), strong trends in area and range can not be expected to have occurred in the recent past. However, the habitat is recovering naturally due to aging of stands adjacent to qualifying habitat and to expanding beech. Nowadays, beech succesfully colonizes even drift sand afforestations. At the same time, holly spreads into almost any forest type (mostly from gardens, cemetries etc.), facilitated by climate warming. Strong postive trends do occur with respect to structure. After the abandonment of coppice as forestry measure (see above), beech is expanding rapidly by natural rejuvenation (initially from lane and road side plantings) at the expense of oak which shows hardly any natural regeneration in closed woodland. As a result of these trends, most (former coppice) stands become denser, darker and with a reduced contribution of oak and herb layer cover. Coarse dead wood, both standing and lying, increases steadily (Schelhaas *et al.* 2014). Associated species such as woodpeckers, nuthatch, dead wood fungi and bryophytes show positive trends according to national red lists (and see Boele *et al.* 2016). The absence of particular characteristic species (mainly invertebrates, fungi and bryophytes) is mainly caused by poor habitat quality and dispersal constraints rather than by insufficient area.

Conclusions

In the historical past, the area of habitat type 9120 had already been diminished to a large extent due to deforestation. After the Middle Ages, scattered woodland remnants have been used mostly as oak coppice or degraded further to heathland. In the recent past, coppicing was abandoned and ancient woodland was converted to high forest or replaced by productive coniferous forest. After about 1980 a more nature-based forst management became accepted at the benefit of native deciduous tree species.

A strong negative trend in area is associated with changes in land use going back to the Middle Ages. In the recent past, the habitat shows natural recovering of some area and certainly of structure & function. The natural range appears rather intact but the current distribution within this range is highly fragmented. Apparently, the habitat is recovering naturally after a deep low.

Step 2.1 - FRA assessment

The previous conclusion requires a FRA assessment according to step 3.1 of the step-wise approach (Natural recovering habitat) for which a reference-based method is inappropriate. Instead, an area-based method is applied. In order to allow the development and persistence of high quality habitat at the scale of stand dynamics on a long-term basis (considering its functioning at the macro-habitat level; see 1.2), more or less continous areas of habitat are needed of at least 125-200 ha, based on a MSA of a 25-40 ha for native beach-oak forest (Koop & Van der Werf 1995) and a fivefold upscaling to allow for large-scale disturbances. This requirement is only (though hardly) met in a few forest complexes within the Veluwe Natura 2000-site (in the centre of the country) such as the Speulderbos/Sprielderbos and Kroondomein (Bijlsma 2014). In order to account for geographical (edaphic and climatological) variation of the habitat across the country, as well as risk spreading, at least two other sites must have continuous areas of habitat about 125-200 ha. This is ecologically feasible only in sites with a substantial area of qualifying ‘ancient woodland’ (see definition above) and this only occurs in the ‘push moraine-landscapes’ (as distinct from ‘cover sand-landscapes’) within the physical geographic region Higher sandy soils, in particular in the Natura 2000-sites ‘Landgoederen Oldenzaal’ (near eastern border with Germany/Lower Saxony) and ‘Sint Jansberg’ (part of the large ancient woodland Nederrijkswald continued in Germany/Westphalia as ‘Reichswald’), currently with 113 and 84 ha of 9120 habitat respectively (Janssen *et al.* 2014). Therefore, an additional area of about 150 ha of habitat is needed. Because this extra area contributes only to the FRA when it is developed in specific sites, we don’t assess the FRA as current value + extra area but use an operator.

Conclusions

FRA > CV (requiring about 150 ha extra in specific sites).

Step 2.2 - FRR assessment

Since remnants of qualifying old woodland still occur throughout the physical geographic region Higher sandy soils, i.e. the potential range, the current range of 9120 is sufficiently large.

Conclusions

FRR=15,600 km^2 (Bijlsma 2014).

References

Bijlsma, R.J., G.J. van Dorland, D. Bal & J.A.M. Janssen (2010). Oude bosgroeiplaatsen. Een referentiebestand voor het karteren van de habitattypen Beuken-eikenbossen met hulst en Oude eikenbossen. Alterra-rapport 1967, Wageningen.

Bijlsma, R.J. (2014). H9120 Beuken-eikenbossen met hulst. In Bijlsma, R.J., J.A.M. Janssen, E.J. Weeda & J.H.J. Schaminée (2014). Gunstige referentiewaarden voor oppervlakte en verspreidingsgebied van Natura 2000-habitattypen in Nederland. WOt-rapport 125, Wettelijke Onderzoekstaken Natuur & Milieu, Wageningen: 174-176 [Favourable reference values for area and range of Natura 2000 habitat types in the Netherlands. Wageningen, Statutory Research Tasks Unit for Nature and the Environment, Wageningen UR. WOt-report 125].

Boele, A., J. van Bruggen, F. Hustings, K. Koffijberg, J.W. Vergeer & T. van der Meij (2016). Broedvogels in Nederland in 2014. Sovon-rapport 2016/04. Sovon vogelonderzoek Nederland, Nijmegen [Breeding birds in the Netherlands in 2014]

Hahn, K., J. Emborg, L. Vesterdal, S. Christensen, R.H.W. Bradshaw, K. Raulund-Rasmussen & J.B. Larsen (2007). Natural forest stand dynamics in time and space - synthesis of research in Suserup Skov, Denmark and perspectives for forest management. Ecological Bulletins 52: 183-194.

Janssen, J.A.M., E.J. Weeda, P. Schipper, R.J. Bijlsma, J.H.J. Schaminée, G. Arts, Ch. Deerenberg, O. Bos & R. Jak (2014). Habitattypen in Natura 2000-gebieden. Beoordeling van oppervlakte, representativiteit en behoudsstatus in de Standard Data Forms (SDFs). Technical report 8, Wettelijke Onderzoekstaken Natuur & Milieu, Wageningen.

Kaplan, J.O., K.M. Krumhardt & N. Zimmermann (2009). The prehistoric and preindustrial deforestation of Europe. Quaternary Science Reviews 28: 3016–3034.

Koop, H. & S. van der Werf (1995). Natuurlijke bosgemeenschappen A-locaties en boscomplexen. Achtergronddocument bij de Ecosysteemvisie Bos. IBN-rapport 162, Wageningen.

Pott, R. (1990). Die nacheiszeitliche Ausbreitung und heutige pflanzensoziologische Stellung von Ilex aquifolium L. Tuexenia 10: 497–512.

Schelhaas, M.J., A.P.P.M. Clerkx, W.P. Daamen, J.F. Oldenburger, G. Velema en, P. Schnitger, H. Schoonderwoerd & H. Kramer (2014). Zesde Nederlandse Bosinventarisatie; Methoden en basisresultaten. Alterra-rapport 2545. Wageningen, Alterra Wageningen UR (University & Research centre) [Sixth Dutch forest inventory; methods and basic results]

Walther, G.-R., S. Berger & M.T. Sykes (2005). An ecological ‘footprint’ of climate change. Proc. R. Soc. B 272: 1427–1432.

9.6 *Quercus suber* forests (9330) in Italy

Emiliano Agrillo & Fabio Attorre

Step 1.1 - Ecology of the habitat
On the Italian territory, habitat 9330 occurs mainly along the Tyrrhenian coast, on the major islands and in a restricted part of the Adriatic coast (Apulia). The distribution is limited to the coastal belt and, in some cases, reaches also the hills of subcoastal zones, on acidophilous sandy soils, with sufficient moisture content to compensate for the summer drought typical of the thermo-Mediterranean climate.

The habitat is characterized by evergreen woods, in which *Quercus suber* (cork oak) is the dominant species. Cork oak is an evergreen broadleaved tree, slow-growing and long-lived (200 years or more). It usually grows up to 20 m in height, but can reach 25 m and a diameter of 1.5 m. It is a monoecious, wind-pollinated species. The flowers appear from spring onwards throughout the summer, and can give rise to both annual and biennial acorns. Acorns are 2-3 cm in length in a fairly deep cup with elongated scales. The bark is thick and fissured and the underlying trunk has a rich red colour. Cork oak needs an average temperature of around 15 °C to thrive and cannot tolerate temperatures below -10 °C, which determines its northern and altitudinal range: most cork oak forests are found below 800 m altitude. It prefers sandy and lightly structured soils. It has a number of adaptations for growing in a warm and dry climate: it has an extensive and deep root system allowing it to cope well with drought, and it can also close the stomata on its leaves to restrict water loss. However, it can also grow in areas of high rainfall (for example in north-west Portugal with average annual rainfall of 2400 mm). Cork oak is well adapted to cope with fire, an ever-present hazard in Mediterranean regions, as its thick bark protects the tree which is capable to re-sprout from the stem after fire damage - the only European tree species with this capability.

In the 9330 habitat, cork oak is often codominant with other evergreen oaks (e.g. *Quercus ilex ilex*); locally, other deciduous species (*Quercus pubescens* s.l., *Quercus cerris*, *Quercus frainetto*, *Fraxinus ornus*) contribute to the tree layer. The undergrowth is mainly composed of Mediterranean maquis species. *Rubia peregrina*, *Smilax aspera*, *Erica arborea*, *Arbutus unedo*, *Rubus ulmifolius* are the most frequent species of the shrub layer, and *Ruscus aculeatus*, *Rosa sempervirens*, *Brachypodium sylvaticum* and *Cyclamen repandum* in the herb layer.

The structure of undisturbed woods can show a continuous tree layer. If disturbances are too intensive, the habitat can degrade into a savanna-like community, with an open tree layer and an increase of heliophile species and pyrophytes (e.g. *Ampelodesmos mauritanicus*, *Cistus salviifolius*).

Regarding its ecological functioning, the habitat 9330 is especially used by fauna restricted to the termophilous vegetation belt along the Italian coast, such as xylophagous beetles (e.g. C*arabus famini, Cerambix velutinus, Cerambyx welensii, C. cerdo e C. miles*) and Lepidoptera (e.g. L*imenitis reducta, Zerynthia polyxena, Lasiocampa quercus, Gastropacha quercifolia*).

Step 1.2 - Spatial scale of functioning
These woody communities have always been subjected to exploitation and other human influences. E.g. fire can induce the degradation of the cork oak wood into savannah-like stands, maquis or garrigue communities in a relative short time (decades).

Quercus suber is more hygrophilous than other evergreen oaks. Since climate changes toward drier conditions and cork exploitation declines, this favours succession to *Quercus ilex* dominated or co-dominated woods.

Conclusions

The habitat type functions at the meso-habitat level (category 2a).

Step 1.3 - Historical perspective: what happened to the habitat?

The distribution of the habitat (Figure 9.6.1) is determined by the historical natural and human influenced distribution of *Qurecus suber*. In the historical past, habitat type 9330 was more widespread than nowadays. The current extent of cork-oak woodlands is the remnant part of the ancient and native Mediterranean forests.

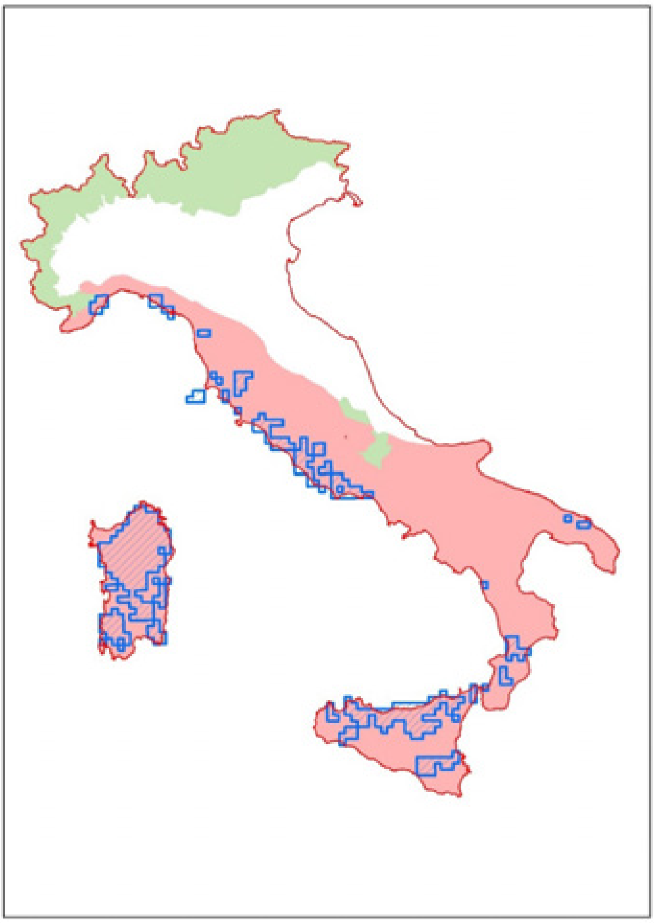

Figure 9.6.1 *Current distribution (in blue grids of10x10 km squares) of habitat 9330 in Italy as reported in 2013 (biogeographical regions: green ALP; white CON and red MED)*

Palynological analyses show that *Q. suber* pollen reached maximum values during the early and middle Holocene. A Late Glacial expansion of *Q. suber* can be seen in pollen diagrams from Morocco. On the other hand, economic activities in the late Holocene could have maintained and even increased the range of the species in particular areas such as the western Iberian Peninsula and Italian territories. On Corsica *Q. suber* and *Q. ilex* often replace deciduous oak forests after the first evidence of human activities. In north-west Tunisia, an anthropogenically induced succession can be demonstrated towards *Q. suber* and *Pinus pinaster* and its fire-derived understorey with *Phillyrea*. The antiquity of human influence ensures that there are no pristine cork oak forests in the Iberian Peninsula and elsewhere. In the absence of human intervention, *Q. suber* would be a subordinate component, sharing the arboreal stratum with other sclerophyllous and deciduous *Quercus* and *Pinus pinaster*. Most dense monospecific forests of cork oak must be a result of human management.

The cork-oak communities have been exploited by humans, that managed them as savanna-like woodlands (30-60 trees per hectar) and created open woodlands for the agro-silvo-pastoralism activities including the harvesting of the cork bark and of acorns for feeding pigs. This kind of exploitation, through clearance and regrowth, as well as the natural disturbance of fires, affected both the structure and the species composition of these woodlands and favoured their expansion. The abandonment of these activities likely turn the *Quercus suber* woodlands into *Quercus ilex* dominated woods. In some sites (for example, Apulia) there has been a remarkable decline in cork oak woods since 1950, due to their conversion into farmlands. In summary, the absence of disturbance factors, such as fire and human management (cork harvesting) could reduce the presence of *Quercus suber* woodlands.

The European Red List of Habitats reported that in Europe the area of the habitat G2.1 Mediterranean evergreen *Quercus* woodland is stable or increasing[13]. Nevertheless it also suggests that the subtype of *Quercus suber* dominated woodlands should be taken into account for conservation, because it appears to be declining in part of the range (Italy). In fact in Italy, even in the absence of quantitative historical data, a decrease in the total distribution area of circa 5% has been observed by the territorial experts. While in agricultural landscapes the area of *Q. suber* woodland is increasing, the natural forests with cork oak are decreasing.

Step 1.4 - Analysis of distribution and trends

Historical data are fragmented and estimates of historical areas and trends are still missing. Negative trends at the biogeographical level can be inferred from distribution data of *Quercus suber* as mentioned in the previous sections. There is no signal for natural recovering of the habitat type. The economic value (cork bark, acorns) of *Quercus suber* and the selective influence of some factors (such as fire) favoured its spread. The gradual abandonment of agro-silvo pastoral practices and climatic and other land-use changes are limiting the distribution of *Quercus suber* and reducing the extent of the habitat type 9330.

Conclusions

The area of habitat 9330 was reduced in Italy by circa 5% in the last 50 years. This decline is not due to changes in ecological requirements but to climatic and land-use changes, especially the gradual abandonment of traditional agro-silvo pastoral practices. There is no signal for natural ongoing recovery.

Step 2.1 - FRA assessement

The habitat 9330 was reduced by ca 5% and is still in decline (see 1.4). In 2013 the surface area of 2,385 km^2 was reported as inadequate with a decreasing trend in MED. The RefValue obtained by spatial modelling (see Supplement) is somewhat larger but has the same order of magnitude, so RV1 (the distance to reference value) is small.

Conclusions

FRA > 2,385 km^2.

Step 2.2 - FRR assessement

The FRR reported in 2013 is 41,800 km^2, completely included in the MED region, with a decreasing short-term trend. Given the qualitative criteria applied for estimation only an operator can be used for the FRR assessement.

Conclusions

FRR > 41,800 km^2.

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[13] https://forum.eionet.europa.eu/european-red-list-habitats/library/terrestrial-habitats/g.-forests/g2.1-mediterranean-evergreen-quercus-woodland

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References

Agrillo, E., Alessi, N., Massimi, M., Spada, F., De Sanctis, M., Francesconi, F., Cambria, V.E. & Attorre F. 2016. Nationwide Vegetation-Plots Database – Sapienza University of Rome: State of the art, basic figures and future perspectives. Phytocoenologia DOI: http://dx.doi.org/10.1127/phyto/2016/0139.

Agrillo, E., N. Alessi, B. Jiménez-Alfaro, L. Casella, P. Angelini, O. Argagnon, G. Crespo, F. Fernández-González, T. Monteiro-Henriques, C.S. Neto & F. Attorre. 2018. The use of large databases to characterize habitat types: the case of Quercus suber woodlands in Europe. Rend. Fis. Acc. Lincei https://doi.org/10.1007/s12210-018-0703-x.

Angelini, P., Casella, L., Grignetti, A., & Genovesi, P. (2016). Manuali per il monitoraggio di specifiee habitat di interesse comunitario (Direttiva 92/43/CEE) in Italia: habitat. ISPRA, Serie Manuali e linee guida, 142(2016), 280. (http://www.isprambiente.gov.it/it/pubblicazioni/manuali-e-linee-guida/manuali-per-il-monitoraggio-di-specie-e-habitat-di-interesse-comunitario-direttiva-92-43-cee-in-italia-habitat).

Bacchetta G., Bagella S., Biondi E., Farris E., Filigheddu R., Mossa L., 2004. A contribution to the knowledge of the order Quercetalia ilicis Br.-Bl. ex Molinier 1934 of Sardinia. Fitosociologia 41 (1): 29-51.

Biondi E., Casavecchia S., Gigante D., 2003. Contribution to the syntaxonomic knowledge of the Quercus ilex L. woods of the Central European Mediterranean Basin. Fitosociologia 40 (1): 129-156.

Biondi E., Casavecchia S., Guerra V., Medagli P., Beccarisi L., Zuccarello V., 2004. A contribution towards the knowledge of semideciduous and evergreen woods of Apulia (south-eastern Italy). Fitosociologia 41 (1): 3-28.

Burrascano S, Casavecchia S, Copiz R, Del Vico E, Galdenzi D, Gigante D, Lasen C, Spampinato G, Venanzoni R, Zivkovic L (2009a) Italian interpretation Manual of the habitats (92/43/EEC Directive). Ministero dell’Ambiente e della Tutela del Territorio e del Mare. Http://vnr.unipg.it/habitat/.

Carrión J. S., Parra I., Navarro C., & Munuera M., 2000. Past distribution and ecology of the cork oak (Quercus suber) in the Iberian Peninsula: a pollen-analytical approach. Diversity and Distributions, 6(1), 29-44.

Casella L., Bianco P.M., Angelini P & Morroni E., 2012. Italian National Vegetation Database (BVN/ISPRA). In: Vegetation databases for the 21st century. Biodiversity & Ecology, 4. ISSN: 1613-9801.

Chytrý, M., Hennekens, S. M., Jiménez-Alfaro, B., Knollová, I., Dengler, J., Jansen, F., ... & Ambarlı, D. (2016). European Vegetation Archive (EVA): an integrated database of European vegetation plots. Applied vegetation science, 19(1), 173-180.

EEA - Conservation status of habitat types and species (“Italian Habitats Directive Reporting” - Article 17, Habitats Directive 92/43/EEC) - https://www.eea.europa.eu/data-and-maps/data/article-17-database-habitats-directive-92-43-eec-1.

ISPRA, Corine land cover 2006 IV livello. Dati della Rete del sistema Informativo Nazionale Ambientale – SINAnet.

Janssen, J. A. M., Rodwell, J., Nieto, A., Gubbay, S., Haynes, T., Nabuurs, G. J., & Schaminee, J. H. J. (2014). Red list of European Habitats Project. In Book of abstracts of 23rd Annual Meeting of European Vegetation Survey EVS (pp. 220-220). (Eunis terrestrial Habitat https://forum.eionet.europa.eu/european-red-list-habitats/library/terrestrial-habitats).

Testi A., Lucattini C., 1994. Contribution to the syntaxonomic knowledge of Quercus suber woodlands of Latium. Atti dell’Accademia Nazionale dei Lincei - Serie IX - V (III): 247-259.

Supplement: Modelling of the potential range

To estimate the potential of recovery of the natural range of habitat type 9330, spatial modelling has been carried out, through the following steps:

- Selection of habitat type plots (see Figure 9.6.2A). The vegetation plots with characteristic/diagnostic species (i.e. by comparing the national and European habitat interpretation manual) were selected in the available national georeferenced vegetation databases (e.g. BVN-ISPRA and VDP-SAPIENZA). A further selection was made taking into account the structure of the habitat (relative cover of the diagnostic/characteristic species (e.g. equal or higher than 50% with respect to other tree species cover).
- Spatial modelling. Spatial predictive modelling was performed by Maxent, to estimate the geographic natural range of the habitat and the correlation with major environmental factors (e.g. seasonality and annual mean temperature, seasonality and annual precipitation, terrain elevation, slope and aspect).
- The preliminary result is a suitability map for the habitat 9330 across the Italian territory (i.e. "wider potential range"). The lower suitability values were excluded by using a threshold value provided by Maxent. Furthermore, territories occupied by patches of land that are not compatible with the existence of a natural habitat (i.e. urban areas, intensive and extensive agricultural area etc.) or patches not compatible with the habitat under examination (i.e. dunes, rivers etc.) were excluded.

The final result is a "potential range" map of habitat 9330 (see Figure 9.6.2B) of 57,200 km^2 (expressed as 10x10 km squares).

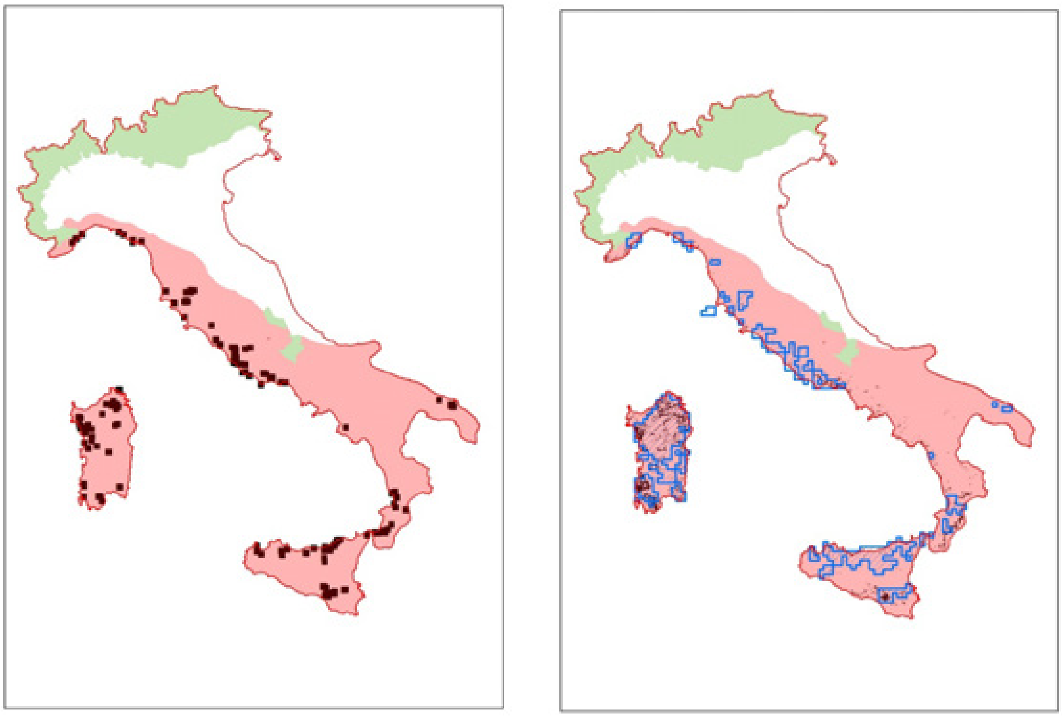

Figure 9.6.2 *Modelling of the potential range of 9330 in Italy. A (left). Selected plots for habitat 9330 on the map of the biogeographical regions (ALP green, CON white, MED red). B (right). Resulting potential range of habitat 9330 in black with current distribution outlined in blue*

9.7 Alpine *Larix decidua* and/or *Pinus cembra* forests (9420) in Italy

Emiliano Agrillo & Fabio Attorre

Step 1.1 - Ecology of the habitat
In Italy habitat 9420 includes subalpine or sometimes alpine forests with prevalence of *Larix decidua* and/or *Pinus cembra*, constituting pure or mixed formations, sometimes associated with *Picea abies* or *Pinus uncinata*. In the upper montane belt (1400-1800 m) in some slopes a combination of *Pinus sylvestris, P. cembra, P. mugo, Picea abies* and *Larix decidua* can be found. A subalpine *Larix decidua-Pinus cembra* belt (m 1800-2000) is also present. Here *Picea abies* is absent or rare, while *Abies alba* is well represented in the *Rhododendron ferrugineum* undergrowth, with or without *Alnus viridis*. In the Lombard south-alpine sector (Alpi Orobie and Adamello group), *Pinus cembra* formations are often found on grasslands of *Festuca scabriculmis* associated with a shrub layer with *Juniperus nana*. In these situations, *Picea excelsa* is often present.

By definition these forests are highly heterogeneous in composition, including *Larix decidua* or *Pinus cembra* dominated woods and mixed woods with different types of understory. A high anthropogenic or natural disturbance tends to favour *Larix decidua*, while a low disturbance, especially in continental conditions, is more favourable to *Pinus cembra*. Moreover, *Pinus cembra* in some districts of the Alps is intentionally eliminated as poorly suited for pasture. Grazing activities in particular have a large influence on distribution and composition of the habitat: extensive grazing maintains *Larix decidua*. In order to favour *Pinus cembra* (where continental conditions allow it) it is best to avoid extensive pasture.

Typical species are *Larix decidua* and *Pinus cembra*. Two subtypes can be distinguished: one on silicate substrates, with *Picea abies*, spread from the western to the oriental Alps, mainly in the inner chains, with a relatively poor flora: *Vaccinium myrtillus, Rhododendron ferrugineum, Calamagrostis villosa, Luzula albida*. The other one includes subalpine formations on carbonate substrates, also mixed with *Picea abies*, in the central and eastern Alps, but with a rich undergrowth: *Erica herbacea, Polygala chamaebuxus, Rhododendron hirsutum, Pinus mugo*. Bryophytes and lichens, growing on trees, but also on other substrates (rocks, dead wood, tree trunks), are of great importance for the biodiversity of this habitat. The habitat type 9420 is relevant for a fauna restricted to alpine conditions, such as the resident grouse species western capercaillie (*Tetrao urogallus*), black grouse (*Lyrurus tetrix*) and rock ptarmigan (*Lagopus mutus*), as well as xylophagous beetles (e.g. *Stephanopachys substriatus*).

Step 1.2 - Spatial scale of functioning
This forest habitat is characterized by dynamic changes in structure, composition and dominant tree species. The occurrence of *Pinus cembra* indicates situations closer to natural conditions; grazing and human activities, apart from natural phenomena such as snow and debris erosion, increase the competitiveness of *Larix decidua*, which can form stable communities especially in the eastern Alps. *Larix decidua* and *Pinus cembra* can colonize rocky slopes as well as debris and massive stabilized boulders. The minimum area for functioning ranges from several to 10s of hectares, dependent on specific locations that are part of an abiotic mosaic at a larger scale.

Conclusions

The habitat type functions at the meso-habitat level (category 2a).

Step 1.3 - Historical perspective: what happened to the habitat?
This forest habitat is characterized by dynamic changes in structure, composition and dominating trees, where *Larix* grows in mixture with *Picea*, in isolated pure stands, or as solitary trees on meadows at 1000–2000 m elevation. No quantitative data on historical range and area are known.

Nevertheless, this complexity in structure could favour the overall conservation of the habitat type. Whereas pure stands of *Larix* are favoured by intensive grazing, mixed or *Pinus cembra* dominated woods are favoured by abandonment, this could mean that changes in traditional use (abandonment/ intensive grazing) do not affect significantly the area in the long term. Overall, even though no exact data is available, the habitat used to be more widespread in the historical past and is slowly increasing during the last decades.

Figure 9.7.1 *Current distribution (in blue grids of 10x10 km squares) of the habitat type 9420 in Italy as reported in 2013 (biogeographical regions: green ALP; white CON and red MED)*

Step 1.4 - Analysis of distribution and trends

There is a positive trend in area due to natural recovering of this habitat type (see step 1.3). Since none of the characteristic species or structures are in bad condition, there is no indication that the habitat is not viable in the long term regarding its function & structure.

Conclusions

Changes in land use apparently lead to its natural recovering during the past decades after a historical decline. The abandonment of pastures and meadows is favouring the recolonization of vast areas, mainly by Larix decidua. The range is likely to be already stable over several decades.

Step 2.1 - FRA assessment

The current value is considered as FRA, because of the increase over the last decades and the absence of indicators of non-viable functioning or structure. For this the only available quantitative estimate is chosen, from the 2013-report, based on partial data (regional maps) with some extrapolation.

Conclusions

FRA can be considered equal to the area reported in 2013: FRA = 3,108 km^2.

Step 2.2 - FRR assessment

The current range (Figure 9.7.1) is large enough to contain the current FRA and therefore is considered as the FRR. Enlargement of the range seems possible (see Supplement) but is not required for a viable structure and function.

Conclusions

FRR= 44,100 km^2.

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We would like to thank Dr. Laura Casella and Dr. Pierangela Angelini (Italian National Institute for Environmental Protection and Research - ISPRA) for the careful reading and useful suggestions.

References

Agrillo, E., Alessi, N., Massimi, M., Spada, F., De Sanctis, M., Francesconi, F., Cambria, V.E. & Attorre F. 2016. Nationwide Vegetation-Plots Database – Sapienza University of Rome: State of the art, basic figures and future perspectives. Phytocoenologia DOI: http://dx.doi.org/10.1127/phyto/2016/0139.

Andreis C., S. Armiraglio, M. Caccianiga, B. Cerabolini. La vegetazione forestale dei *Piceetalia excelsae* Pawl. in Pawl. *et al.* 1928 nelle Alpi Lombarde. Fitosociologia (in stampa).

Angelini, P., Casella, L., Grignetti, A., & Genovesi, P. (2016). Manuali per il monitoraggio di specifiee habitat di interesse comunitario (Direttiva 92/43/CEE) in Italia: habitat. ISPRA, Serie Manuali e linee guida, 142(2016), 280. (http://www.isprambiente.gov.it/it/pubblicazioni/manuali-e-linee-guida/manuali-per-il-monitoraggio-di-specie-e-habitat-di-interesse-comunitario-direttiva-92-43-cee-in-italia-habitat).

Burrascano S, Casavecchia S, Copiz R, Del Vico E, Galdenzi D, Gigante D, Lasen C, Spampinato G, Venanzoni R, Zivkovic L (2009a) Italian interpretation Manual of the habitats (92/43/EEC Directive). Ministero dell'Ambiente e della Tutela del Territorio e del Mare. Http://vnr.unipg.it/habitat/.

Casella L, Bianco P.M., Angelini P & Morroni E., 2012. Italian National Vegetation Database (BVN/ISPRA). In: Vegetation databases for the 21st century. Biodiversity & Ecology, 4. ISSN: 1613-9801.

Chytrý, M., Hennekens, S. M., Jiménez-Alfaro, B., Knollová, I., Dengler, J., Jansen, F., ... & Ambarlı, D. (2016). European Vegetation Archive (EVA): an integrated database of European vegetation plots. Applied vegetation science, 19(1), 173-180.

EEA - Conservation status of habitat types and species ("Italian Habitats Directive Reporting" - Article 17, Habitats Directive 92/43/EEC) - https://www.eea.europa.eu/data-and-maps/data/article-17-database-habitats-directive-92-43-eec-1.

Filipello S., Sartori F., Vittadini M., 1981. Le associazioni del cembro nel versante meridionale dell'arco alpino, 2: La vegetazione, aspetti forestali. Atti Ist. Bot. Lab. Critt. Univ. Pavia, s. 6, 14: 1 - 48.

ISPRA, Corine land cover 2006 IV livello. Dati della Rete del sistema Informativo Nazionale Ambientale – SINAnet.

Sburlino G., Lasen C., Buffa G., Gamper U., 2006. Sintassonomia e nomenclatura delle comunità forestali a *Pinus cembra* L. delle Alpi italiane. Fitosociologia, 43 (2): 3-20.

Supplement: Modelling of the potential range

To estimate the potential of recovery of the natural range of habitat type 9420, spatial modelling has been carried out, through the following steps:

- Selection of habitat type plots (see Figure 9.7.2A). The vegetation plots with characteristic/diagnostic species (i.e. by comparing the national and European habitat interpretation manual) were selected in the available national georeferenced vegetation databases (e.g. BVN-ISPRA and VDP-SAPIENZA). A further selection was made taking into account the structure of the habitat (relative cover of the diagnostic/characteristic species (e.g. equal or higher than 50% with respect to other tree species cover).
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- The preliminary result is a suitability map for the habitat 9420 across the Italian territory (i.e. "wider potential range"). The lower suitability values were excluded by using a threshold value provided by Maxent. Furthermore, territories occupied by patches of land that are not compatible with the existence of a natural habitat (i.e. urban areas, intensive and extensive agricultural area etc.) or patches not compatible with the habitat under examination (i.e. dunes, rivers etc.) were excluded.

The final result is a "potential range" map of habitat 9420 (see Figure 9.7.2B. 2b) of 48,000 km^2 (expressed as 10x10 km squares). This potential natural range is circa 10% larger than the current range according to the 2013 report (44,100 km^2).

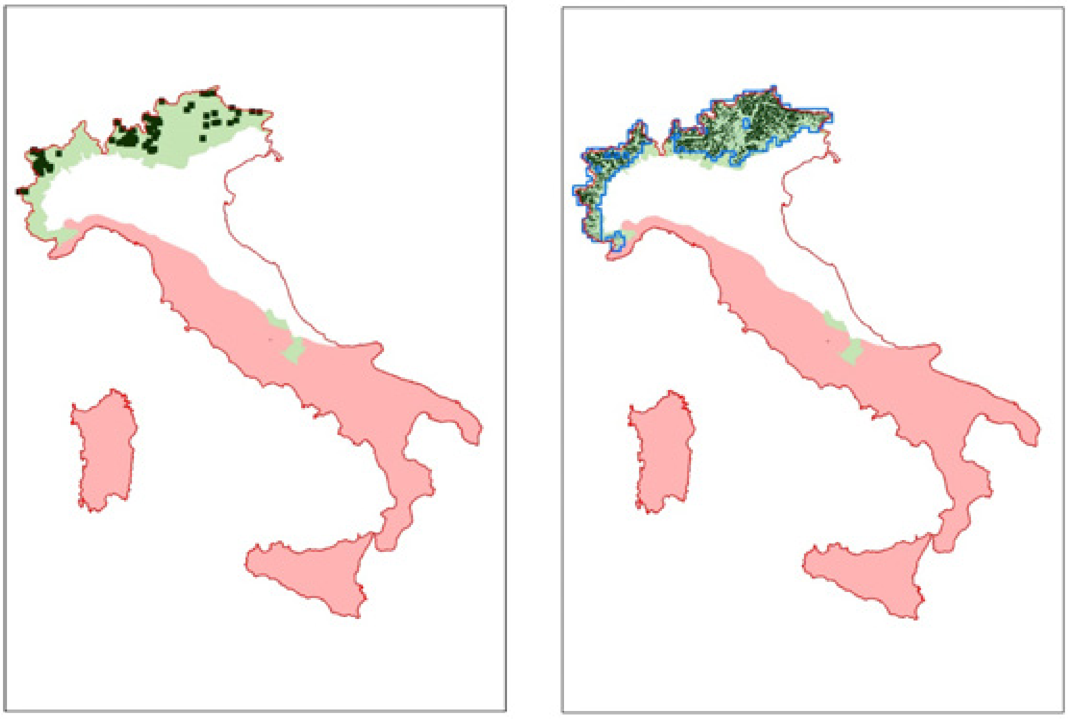

Figure 9.7.2 *Modelling of the potential range of 9420 in Italy. A (left). Selected plots for habitat 9420 on the map of the biogeographical regions (ALP green, CON white, MED red). B (right). Resulting potential range of habitat 9420 in black with current distribution outlined in blue*

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