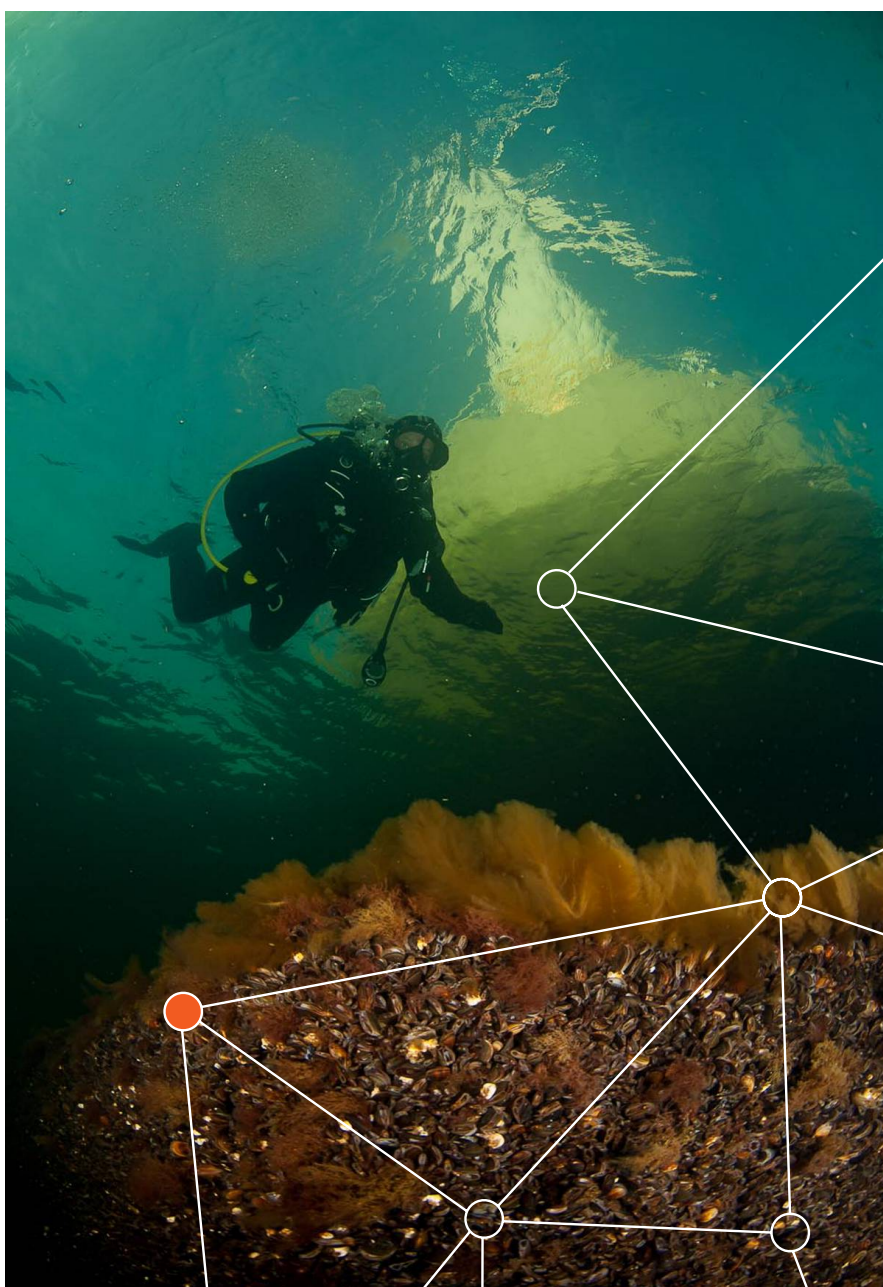


Data-limited diadromous species – review of European status

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Data-limited diadromous species – review of European status

Karen Wilson • Lari Veneranta

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Foreword

ICES Working Group on Data-Limited Diadromous Species (WGDAM) is a subgroup of ICES Working Group on the Science Requirements to Support Conservation, Restoration and Management of Diadromous Species (WGRECORDS). For a full list of members, see Section 6. The group began working in 2015, with the primary task of updating knowledge of the status and distribution of poorly understood diadromous fish species. Diadromous fish species are those that have separate feeding and reproduction areas in saline and freshwater and migrate between them.

In 2005, ICES Diadromous Fish Committee (SGSDFS) reported on the status and distribution of poorly understood diadromous species (ICES, 2005). Since the 2005 report, there have been increasing legal drivers to protect and restore these species, mainly for biodiversity reasons. Many diadromous fish species have threatened status and suffer from environmental degradation and human-induced changes, especially in reproduction areas. The legal drivers, and the associated science, have further highlighted knowledge gaps in the biology of these species. Pressure from development in freshwater, transitional, and marine zones continues to threaten the life cycle of these species. More scientific information is required for these data-limited diadromous species.

This report (i) indicates specific cases where changes in the status of diadromous species are occurring which are directly related to human impact and ecosystem changes, and (ii) reviews the current status of selected species based on existing national and international criteria. All data-limited diadromous species are not covered in this report. Therefore, it should only be used as an initial guidance for future work to cover possible data and management gaps. This report focuses mainly on species in the European area, and covers countries and species represented by WGDAM members.



One week old anadromous whitefish (*Coregonus lavaretus*) from River Perho in western Finland.

The main threats to diadromous fish are the same as in the previous reporting period in 2005, including migration barriers (e.g. dams), river construction, local inputs to rivers, lagoons, and estuaries (pollution, eutrophication, acidification), habitat loss, and overfishing. Many diadromous species are still in great difficulty, including the European sturgeon, allis and twaite shads, European eel, river lamprey, natural stocks of migratory whitefish, and the coastal grayling in the Baltic Sea. Differences between areas and populations are large, e.g. sea trout is in great danger in some areas, but not

in others. Species with reproduction cycles dependent on rivers are threatened as a result of high human pressure applied to coastal areas and riverine habitats, including incompatible land use, e.g. for the purposes of forestry and agriculture.

Since the previous reporting period, more data has been collected, and knowledge has increased for some species, particularly those that have been the focus of ICES Working Groups (eels, lampreys, shads, and sea trout). This report highlights the need for better data on other migratory species living at the transition of fresh and marine environments, such as unique populations of thin lip grey mullet, smelt, whitefish, and coastal grayling.

Major anomalies or changes to the state of the ecosystem

- Continued degradation of most data-limited diadromous fish habitats, specifically estuaries, and access to rivers or estuaries which are reproduction habitats.
- High fishing pressure continues to impact commercially important and by-catch species, especially in the Baltic Sea region. Climate change is predicted to change river flow and timing of floods, which can be a potential threat for reproduction.
- Problems in reproduction have led to weakened populations. However, effective restoration attempts on reproduction areas and fishing restrictions have improved the status of some populations.

Important new methodologies and findings

- With advances in natural tracking techniques and other research technologies (e.g. otolith microchemistry, sidescan sonar, eDNA), it is becoming clear that most diadromous fish species have a greater diversity of life-history strategies than previously assumed. For example, new technologies have made counting fish and identifying migration areas easier than in the past, which, in turn, makes monitoring more tenable.

Emergent issues, challenges, and problems

- Management has become more complicated through the addition of new information on previously unknown life-history strategies that vary by habitat and location. Many species, regional variations, or ecotypes that were previously not considered “diadromous” are now included in this report, because they have migratory behaviour between saline and freshwater areas (e.g. perch, grayling, pike, and flounder).
- Lack of data for many species continues to be a problem for management.
- Barriers to accessing spawning habitats continue to be a problem.
- Many species, or regional variations, are relatively local, making EU-wide or eastern North Atlantic-wide management difficult.
- Diadromous fish species are underrepresented or neglected in the WFD quality criteria.
- Climate change.
- Water abstraction.
- Habitat quality and diversity loss as a result of flood control and inland navigation structures.
- Inefficient habitat protection in candidate areas (Natura 2000).

1 Status of data-limited species

1.1 Background

This report updates the previous SGSDFS (Study Group on the Status of Diadromous Fish Species) baseline report, which was published in 2005 following the DiadFish (Diadromous Fish) initiative¹. SGSDFS report stated that the distribution and status of many diadromous fish species was poorly known. This situation has not changed overall. ICES has advanced the knowledge base of diadromous species with reports and working groups for salmon (ICES, 2015a), sea trout (ICES, 2015b, 2015c), and eels (ICES, ICES 2015e), and workshops for lampreys and shads (ICES, 2014a, 2015d). However, there are still numerous species that may be economically important at least at local or regional scales, but are data-limited and not monitored. In addition, many diadromous species have high importance for biodiversity. Cooperation among European, US, and Canadian research institutes is needed, because species and problems are often similar on both sides of the Atlantic. WGDAM has aimed to strengthen the network, and facilitate cooperation between researchers dealing with diadromous fish.

Diadromous fish use both marine and freshwater environments to achieve their life cycles (McDowall, 1996), and many species face similar problems. The reasons behind the evident reduction in many diadromous fish stocks are multiple, and include: (i) river fragmentation and dams, which may prevent fish from reaching their natural spawning sites (Masters et al., 2006; Béguer et al., 2007; Drouineau et al., 2018); (ii) pollution of rivers and drainage areas, which may also block access to spawning grounds or prevent egg or larval development; and (iii) fishing of commercially important populations, which has had additional effects on the decline of diadromous populations either directly or as a bycatch of other targeted species.

Country borders do not stop human impacts on species that migrate over considerable distances. Hence, besides national programmes, many diadromous species are protected under international conventions. The Bern Convention (Convention on the Conservation of European Wildlife and Natural Habitats), European Habitats Directive, and CITES (Convention on International Trade in Endangered Species of Wild Fauna and Flora) cover the species that are not normally assessed by ICES in any context. This report reviews and collects information on the current status of diadromous species, with the exception of salmon, in the context of international and national classifications. However, some species were omitted from the report owing to the lack of an expert to write the section.

1.2 Geographic area covered by WGDAM

This report focuses on the North Atlantic and Baltic areas. North America (USA and Canada) will be included in a future reporting effort. Specifically, this report includes species from following areas: Mediterranean Sea, North Atlantic, Bay of Biscay, Celtic Sea, North Sea, Norwegian Sea, Baltic Sea, Barents Sea, and Black Sea.

The reporting countries are Belgium, Denmark, Estonia, Finland, France, Germany, Ireland, Italy, Latvia, Lithuania, the Netherlands, Norway, Poland, Portugal, the Russian Federation, Spain, Sweden and UK.

¹ www.diadfish.org

1.3 Criteria for species

The main criteria for species selection in this report was (i) they are diadromous, and (ii) data available to support quantitative stock assessments are limited. Diadromous species are those that migrate between fresh- and seawater during different periods of their life, either to spawn in the sea (catadromous) or in freshwater (anadromous). We have also included some species, particularly in the Baltic region, that migrate between freshwater and low-salinity waters, because these species have similar management issues. Furthermore, species were omitted that have marine–freshwater interactions but fall under the following categories:

- vicarious species: non-diadromous, strictly freshwater species of primarily marine families;
- sporadic species: species that seem to be indifferent to salinity (i.e. euryhaline) and usually occur in estuaries; and
- accidental species: normally marine species that are occasionally and non-predictably caught in inland waters

Most of the diadromous fish species in this report are data-limited across multiple local, regional, and country-level geopolitical borders and ecological biomes. These fish are often locally important, both socially and economically, but do not support valuable commercial or recreational fisheries as a consequence of their small numbers. As a result, less data are collected on stock status, and species become more data-limited. Thus, “data-limited” here refers to the lack of data to support quantitative stock assessment methods, such as statistical catch-at-age analysis or virtual population analysis (Smith *et al.*, 2009), and the inability to conduct large-scale mark–recapture analyses (Pine *et al.*, 2003). The assessments of data-limited diadromous species are often limited to evaluations of uncertain catch and abundance indices.

Species were selected for coverage based on their conservation status or needs, plasticity, and economical importance. DiadFish network (<http://www.diadfish.org/english/FISH.htm>) identified:

- 3 catadromous species: European eel (*Anguilla anguilla*), thin-lipped mullet (*Liza ramada*), and flounder (*Platichthys flesus*).
- 19 anadromous species: sea lamprey (*Petromyzon marinus*), river lamprey (*Lampetra fluviatilis*), European sea sturgeon (*Acipenser sturio*), Adriatic sturgeon (*Acipenser naccarii*), Stellate sturgeon (*Acipenser stellatus*), Russian sturgeon (*Acipenser gueldenstaedtii*), Beluga (*Huso huso*), Atlantic salmon (*Salmo salar*), Black Sea salmon (*Salmo trutta labrax*), sea trout (*Salmo trutta*), Arctic char (*Salvelinus alpinus*), smelt (*Osmerus eperlanus*), houting (*Coregonus lavaretus oxyrinchus*), allis shad (*Alosa alosa*), twaite shad (*Alosa fallax*), and vimba (*Vimba vimba*).

Further details on species selection:

- Salmon was not included in this report because it is well covered in other working groups and assessments.
- Houting, Arctic char, Stellate sturgeon, Russian sturgeon, and Beluga were not included as no available experts on the species were found.
- The Baltic (Atlantic) sturgeon (*Acipenser oxyrinchus*) is included because of on-going restoration efforts in the Baltic Sea.

- Species of the genus *Coregonus* were lumped together in the previous report (ICES, 2005), whereas in this report, *Coregonus lavaretus/maraena* is reported as coastal whitefish.
- Flounder is included, because some special populations have adapted to live in the transition zone of fresh and marine environments.
- Species like perch (*Perca fluviatilis*), pike (*Esox lucius*), and vimba are included, because they have local populations that migrate between feeding and reproduction areas in freshwater estuaries and brackish water environments.

For the purposes of this report, we classify fish species into two groups: (i) those that are completely migratory, and (ii) those that have migratory populations, and are living at the intersection of freshwater and marine environments. This was deemed necessary because recent advances in our knowledge of individual and population level migratory patterns have revealed greater plasticity in life histories than was previously understood.

1.4 Assessments – species categories and status

A number of data-limited diadromous fish species and lampreys are protected under the OSPAR Convention (Convention for the Protection of the Marine Environment of the North-East Atlantic), the European Habitats Directive, CITES, the Bern Convention, IUCN (World Conservation Union), the Bonn Convention (Convention on Migratory Species of Wild Animals (CMS)), HELCOM-IUCN, and national classifications (further details below). With the exception of Atlantic salmon, these diadromous species are not normally assessed by ICES. This report is an expansion and continuum of SGSDFS report (ICES, 2005) and sums up the information on the current status of selected diadromous fish species in the context of classification schemes. The species that are listed in international conventions or classifications are shown in Table 1.1.

1.4.1 OSPAR list of threatened and/or declining species/Texel-Faial

The OSPAR Convention is the current legislative instrument regulating international cooperation on environmental protection in the Northeast Atlantic. It is signed by Belgium, Denmark, the former European Community, Finland, France, Germany, Iceland, Ireland, Luxembourg, the Netherlands, Norway, Portugal, Spain, Sweden, Switzerland, and UK. The OSPAR Biological Diversity and Ecosystems Strategy sets out that the OSPAR Commission will assess which species and habitats need to be protected. Annex V to the OSPAR Convention² – on the Protection and Conservation of the Ecosystems and Biological Diversity of the Maritime Area – was adopted in July 1998 with a strategy on the protection and conservation of the ecosystems and biological diversity of the maritime area.

Annex V to the OSPAR Convention gives the OSPAR Commission the means to institute protective, conservation, restorative, or precautionary measures, consistent with international law, related to specific areas or sites, or to particular species or habitats. The OSPAR List of Threatened and/or Declining Species and Habitats fulfils this commitment, and is based on nominations by contracting parties and observers from the Commission, of species and habitats that they consider to be priorities for protection. Criteria for the Identification of Species and Habitats in need of Protection and their

² https://www.ospar.org/site/assets/files/1169/pages_from_ospar_convention_a5.pdf

Method of Application follow the Texel–Faial criteria³ This list⁴ includes only four migratory fish species identified by WGDAM: European sturgeon, allis shad, European eel, and sea lamprey. Houting (*Coregonus oxyrinchus*), a species reported in earlier SGSDFS (ICES 2005) work is also listed.

The data used in OSPAR assessments have been reviewed by ICES in order to give assurance that their quality is suitable for the purpose for which they are used. The information used has been compiled into a justification report, which is being published separately (Case reports for the OSPAR List of Threatened and/or Declining Species and Habitats (OSPAR 2008).

Further information can be found at: <https://www.ospar.org/>.

1.4.2 European Habitats Directive

The European Habitats Directive ensures the conservation of a wide range of rare, threatened, or endemic animal and plant species. Adopted in 1992, the Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora⁵ aims to promote the maintenance of biodiversity, taking account of economic, social, cultural, and regional requirements. The Directive is the means by which the European Union meets its obligations under the Bern Convention (Section 1.4.4).

If a species is included under this Directive, measures must be taken by individual member states to maintain or restore them to favourable conservation status in their natural range. The Directive-listed habitat types and species are those considered to be the most in need of conservation at a European level. Within the Directive, a collection of these habitats and species have been given priority status because they are considered to be particularly vulnerable and are mainly or exclusively found within the European Union. These sites are designated as Special Areas of Conservation (SACs), and include, for example, the Natura 2000 Network. In SAC areas, member states are obliged to assess the numbers and the level of exploitation of designated species. Three main annexes of the Directive identify the species that require special consideration, and the diadromous species included in each list can be seen in Table 1.1:

- Annex II species (about 900): core areas of the habitat of these species are designated as sites of community importance (SCIs) and are included in the Natura 2000 network. These sites must be managed in accordance with the ecological needs of the species.
- Annex IV species (over 400, including many Annex II species): a strict protection regime must be applied across their entire natural range within the EU, both within and outside Natura 2000 sites.
- Annex V species (over 90): member states must ensure that their exploitation and capture in the wild is compatible with maintaining them in a favourable conservation status.

The reporting cycle of the Habitats Directive is 6 years. 2013 was the first year in the rolling cycle of monitoring and investigation under the Directive. The status of reported species was assessed in member states and submitted to the EU. In reporting,

³ http://jncc.defra.gov.uk/pdf/ANNEX05_Texel_Faial%20criteria.pdf

⁴ <http://www.ospar.org/work-areas/bdc/species-habitats/list-of-threatened-declining-species-habitats>

⁵ <https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:31992L0043>

all member states follow the same procedure, thus enabling a status comparison among countries in Europe.

Further information can be found at: http://ec.europa.eu/environment/nature/legislation/habitatsdirective/index_en.htm

Table 1.1. Diadromous species regulations in the European area: IUCN, Bern Convention, Habitats Directive Fish, and other conventions.

Protected species	European Red List	Habitats Directive annexes	Bern Convention annexes	Other conventions
CLUPEIDAE				
<i>Alosa alosa</i>	LC	II, V	III	OSPAR
<i>Alosa fallax</i>	LC	II, V	III	
<i>Alosa agone</i>	LC ¹			
<i>Alosa algeriensis</i> ¹	DD ¹			
<i>Alosa killarnensis</i> ¹	CR ¹			
COREGONIDAE				
<i>Coregonus albula</i>	LC	V	III	
<i>Coregonus lavaretus</i> *	VU	V	III	HELCOM
<i>Coregonus maraena</i> *	VU	II, IV, V	III	HELCOM
PETROMYZONTIDAE				
<i>Lampetra fluviatilis</i>	LC	II ² , V	III	HELCOM
<i>Petromyzon marinus</i>	LC	II ³	III	OSPAR HELCOM
SALMONIDAE				
<i>Salmo salar</i>	NE	II ⁴ , V	III	OSPAR HELCOM
<i>Thymallus thymallus</i>	LC	V	III	HELCOM
ACIPENSERIDAE				
Adriatic sturgeon (<i>Acipenser naccarii</i>)	CR	II, IV	-	CMS
European sea sturgeon (<i>Acipenser sturio</i>)	CR	II, IV	III	OSPAR CITES
All other Acipenseridae species	CR	V	III, Baltic sturgeon	
ANGUILLIDAE				
European eel (<i>Anguilla anguilla</i>)	CR	-	-	OSPAR CMS CITES HELCOM

DD: data deficient; VU: Vulnerable; NT: Near Threatened; EN: endangered; CR: Critically endangered; RE: regionally extinct; EX: extinct; NE: not evaluated

¹ Newly described species in EU 27 Red List category. ² Annex II: except the Finnish and Swedish populations. ³ Annex II: except the Swedish populations. ⁴ Annex II: except the Finnish populations.

*Often *Coregonus lavaretus* and *Coregonus maraena* describe the same whitefish species.

1.4.3 CITES

CITES is an international agreement among governments. Its aim is to ensure that international trade in specimens of wild animals and plants does not threaten their survival. The species are grouped in the appendices according to how threatened they are by international trade. Currently, only two European diadromous fish species are covered: European sea sturgeon and eel. The shortnose sturgeon, which lives in both Europe and Northern America, is also listed.

Further information can be found at: <https://www.cites.org/eng/app/appendices.php>

1.4.4 Bern Convention

The Bern Convention aims to conserve wild flora and fauna and their natural habitats, especially in cases where conservation requires the cooperation of several states, and to promote such cooperation. It was opened in Bern in 1979 and has been signed by 50 countries and the European Union. Particular emphasis is given to endangered and vulnerable species, including migratory species.

Further information can be found at: <http://www.coe.int/en/web/bern-convention>

1.4.5 Bonn convention/CMS

The Convention on the Conservation of Migratory Species of Wild Animals (CMS, often referred to as Bonn convention) aims to conserve migratory species throughout their range. The convention covers terrestrial, avian and marine species, and it is an international treaty within the United Nations Environment Programme. The convention entered into force in 1983 and it has two appendices that list the species to which the convention applies:

- Appendix I covers endangered migratory species. Signatory states within the range of listed species are required to strictly protect them.
- Appendix II covers migratory species with an unfavourable conservation status, and requires other international agreements for conservation and management. Signatory states within the range of the listed species are encouraged by the convention to conclude such conservation and management agreements.

Currently the convention covers 21 fish species (Actinopterygii), but only two of them are included in this report, European sturgeon (*Acipenser sturio*) and European eel (*Anguilla anguilla*). European sturgeon was added to Appendix I in 2005 and European eel to Appendix II in 2014.

1.4.6 IUCN

The IUCN Red List of Threatened Species provides taxonomic, conservation status, and distribution information on taxa that have been evaluated using the IUCN Red List Categories and Criteria. The assessments are provided by the IUCN through its Species Survival Commission (SSC). This classification has been updated, and assessments and reassessments on the IUCN Red List use the second edition, released in 2012, of the 2001 IUCN Red List Categories and Criteria, version 3.1 (IUCN 2001). The main purpose of the IUCN Red List is to define the status of species that are particularly vulnerable to extinction or extirpation, i.e. species listed as critically endangered (CR), endangered (EN), and vulnerable (VU). However, the Red List also includes information on taxa that (i) are categorized as extinct (EX) or extinct in the wild (EW); (ii) cannot be evaluated because of insufficient information (i.e. are data deficient (DD)); and (iii) are either close to meeting the threatened thresholds, or would be threatened

were it not for an ongoing taxon-specific conservation programmes (i.e. near threatened (NT)).

The European Red List of Freshwater Fishes (Freyhof and Brooks, 2011) lists the status of freshwater fish, including marine migrants, and species from brackish waters that regularly enter freshwater habitats. All diadromous species included in this report are also in the lists, but this report is missing houting.

There has been growing interest among countries in using the IUCN Red List Categories and Criteria for national Red List assessments. Some countries have country-specific Red Data Books where classifications have been carried out using the IUCN categories on a more local scale, including subpopulations. As an example Finland and Sweden include Baltic Sea coastal grayling (*Thymallus thymallus*) in their lists.

Further information can be found at: <http://www.iucnredlist.org/>

1.4.7 HELCOM-IUCN

The HELCOM Red List of Baltic Sea species in danger of becoming extinct (2013) is the first threat assessment for Baltic Sea species that covers all marine mammals, fish, birds, macrophytes (aquatic plants), and benthic invertebrates, and follows the Red List criteria of the IUCN. These species are referenced in HELCOM (2013a). Among diadromous fish species, the list includes river lamprey, sea lamprey, Atlantic sturgeon, European eel, whitefish (*Coregonus maraena*), burbot (*Lota lota*), salmon, sea trout, and grayling.

Further information can be found at: <http://www.helcom.fi/baltic-sea-trends/biodiversity/red-list-of-species>

1.4.8 Examples of other national classifications and protection

UK: UK Wildlife and Countryside Act 1981 (<https://www.legislation.gov.uk/ukpga/1981/69>).

UK: Biodiversity Action (<http://jncc.defra.gov.uk/ukbap>).

USA: Endangered (EN) Species Act (<https://www.fws.gov/international/pdf/esa.pdf>).

Canada: Species at Risk Act (<http://laws-lois.justice.gc.ca/eng/acts/s-15.3/>).

2 Anadromous species

2.1 Allis shad and Twaite shad

2.1.1 Taxonomy

Class:	Actinopterygii
Order:	Clupeiforme
Family:	Clupeidae
Scientific name:	Allis shad - <i>Alosa alosa</i> (Linnaeus, 1758)
	Twaite shad - <i>Alosa fallax</i> (Lacépède, 1803)

Subspecies, variations, synonyms, and taxonomic notes

Alosa fallax spp.

Six subspecies of *Alosa fallax* have been recognized by Quignard and Douchement (1991) based mainly on the number of gillrakers on the first gill arch and geographical location: *Alosa fallax algeriensis* (Regan, 1916), *Alosa fallax fallax* (Lacépède, 1803), *Alosa fallax killarnensis* (Regan, 1916), *Alosa fallax lacustris* (Fatio, 1890), *Alosa fallax nilotica* (Geoffroy Saint-Hilaire, 1808), and *Alosa fallax rhodanensis* (Roule, 1924). However, Kottelat (1997), who critically reviewed the systematics of European freshwater fish, concluded that some of the subspecies identified by Quignard and Douchement (1991) should be reclassified as species based on differences in morphology, habit, life history, and ontogeny: *Alosa fallax algeriensis* as *Alosa algeriensis*, *Alosa fallax killarnensis* as *Alosa killarnensis*, and *Alosa fallax lacustris* as *Alosa agone* (Scopoli, 1786). Further, Kottelat (1997) suggested that, until explicit data were available, the three subspecies *Alosa fallax fallax*, *Alosa fallax nilotica*, and *Alosa fallax rhodanensis* should be classified as *Alosa fallax*.

Considerable confusion still remains, and it is clear that the systematics of *Alosa fallax* spp. need to be addressed, especially as there is a move away from subspecies. However, until the correct nomenclature can be determined, this report refers to the anadromous species of *Alosa fallax* inhabiting the Atlantic seaboard (including the North and Baltic seas) as *Alosa fallax fallax*. For *Alosa fallax*, the following anadromous subspecies will be included in this report: *Alosa fallax algeriensis*, *Alosa fallax fallax*, *Alosa fallax nilotica*, and *Alosa fallax rhodanensis*.

Common names

Alosa alosa

NO: majfisk; SE: majfisker; FI: pilkkusilli; RU: aloza, maiskaya ryba; PL: aloza; DE: maifisch; DK: majsild; NL: Groote meivisch, Elft; EN: allis shad; CY: herlyn; FR: Alose vraie; Pêchon de mai; Abèye, Alôïe; ES: Sábalo; AY: chebel

Alosa fallax

NO: stamsillen; SE: staksillen; FI: täpläsilli; EE: vinträim; LV: Lapreņģe, skalla, palede, Atlantieskaya finta; RU: финта средиземноморская, Sredizemnomorskaya finta; PL: savetha, parposz; DE: finte, elf; DK: stavssild; NL: meivisch, fint; EN: twaite shad; CY: wangen; EN-IE: alose feinte, alose feinte du Rhône, cepa, lojka; péchon de mai; abèye, alôïe, meivis; FR: saboga; ES: savelha, saboga; AY: chbouk, ouarsa; renget El sabawgha-sardine, sardeena, sarboura; chbouka; chbouka; AR: alosa, cheppia; IT: čepa; HR: lloca, kubla; AL: sardelomana; GR: beneklitirsi balığı

2.1.2 Life cycle and migrations

Alosa alosa

Allis shad (*Alosa alosa*) mature at 3–6 years of age and migrate many hundreds of kilometres upstream into their natal river to spawn. Spawning occurs during spring in the main river and major tributaries, in shallow waters, and over gravel substrate (Baglinière *et al.*, 2003). The adults usually die after spawning. Allis shad larvae hatch within 4–5 days, spend summer in freshwater, and migrate to estuaries during autumn. Before they complete their first year of life, they migrate back to the ocean. For more information on the migrations of *A. alosa* see Aprahamian *et al.* (2015).

Alosa fallax

Anadromous populations of twaite shad (*Alosa fallax*) mature at 2–9 years, with the majority of females maturing at age 4–5 years and the males one year earlier (Aprahamian *et al.*, 2003). Nearly all anadromous populations of *A. fallax* are iteroparous, with a large proportion of repeat spawners. However, Moroccan populations in the Sebou and Loukos rivers are semelparous (Sabatié, 1993). Spawning occurs in spring–summer, with the juveniles migrating seaward in autumn. A portion of the one-year-old fish re-enter the estuary in spring before again migrating seaward in autumn (Aprahamian, 1988). For more information on the migrations of *A. fallax* see Aprahamian *et al.* (2015).

2.1.3 Stock structure and population dynamics

This section is confined to characterizing the adult component of the stock. It should be noted that data interpretation is complicated because: (i) there has generally been no correction for gear selectivity, with the exception of the study from Lambert *et al.* (2001); and (ii) different methods operate in different locations, making comparisons between systems and between species difficult.

2.1.3.1 Sex ratio

Alosa alosa

In the Loire (France), the sex ratio of the *Alosa alosa* population over a six-year period varied from 0.84:1 to 4.5:1. In three of the six years, there was no significant difference from a 1:1 ratio; in the other years, there were significantly more males than females ($p < 0.01$) (Mennesson-Boisneau and Boisneau, 1990).

In the Gironde-Garonne-Dordogne (France) system, the sex ratio of fish caught in the estuary (Anon., 1979; Taverny, 1991a) and on the spawning grounds (Cassou-Leins and Cassou-Leins, 1981) showed no significant difference from parity ($p > 0.05$). In contrast, Lambert *et al.* (2001) found that the sex ratio ranged from 0.76:1 to 1.47:1, over a three-year period, with the proportion of females increasing with age. On the River Aulne (France), Acolas *et al.* (2006) observed an overall sex ratio of 0.92:1, with males predominating at the start (1.3:1) and at the end of the migration period (1.2:1) and females during the middle (0.6:1).

No significant difference from a 1:1 ratio was reported from the Portuguese rivers Minho (Mota and Antunes, 2011), Douro (Eiras, 1981), and Lima (Alexandrino, 1996a). In the Oued Sebou (Morocco), Sabatié (1993) studied the sex ratio in four years between 1978 and 1985 and observed the annual sex ratio varied from a preponderance of females (0.5:1) to one year where males dominated (4.8:1).

Alosa fallax fallax

For samples obtained using rod and line from the rivers Wye (Wales), Severn (England), and Barrow (Ireland), no significant difference ($p > 0.05$) from a 1:1 sex ratio was reported by Aprahamian (1982) and O'Maoileidigh (1990). Similar findings were reported for *A. fallax* from the Loire (France; Douchement, 1981), the Gironde-Garonne-Dordogne system (France; Anon., 1979), the Ulla (Spain; Cobo *et al.*, 2010a), and the Lima (Portugal; Alexandrino, 1996a). In contrast, males were more abundant in the Sebou (Morocco), with the sex ratio varying between years from 1.35:1 to 2.15:1 (Sabatié, 1993). Similarly in the Curonian Lagoon (Lithuania), the sex ratio was dominated by males ranging from ca. 3:1 to 5:1 (Repečka, 2003a; Stankus, 2009), whereas females in the Tejo (Portugal) predominated (0.36:1) (Alexandrino, 1996a).

Alosa fallax rhodanensis

In the Rhône (France), the sex ratio has been found to vary between years, changing from parity, to a preponderance of females (0.24:1) and then to a dominance of males (4.15:1) (Douchement, 1981; Le Corre *et al.*, 1997). Samples taken using rod and line showed a much closer agreement to parity, with the ratio ranging from 1.03:1 to 1.28:1 over a four-year period (Barral, 2001). On the Tiber (Italy), D'Ancona (1928) reported a dominance of males, with a sex ratio ranging from 2.44:1 at the start of the migration period (March–April) to 1.30:1 in May and June.

Alosa fallax nilotica

In the River Neretva (Bosnia and Herzegovina), the population was dominated by females, whereas in lakes Bačín (Croatia) and Skadar (Montenegro), the sex ratio was close to 1:1 (Vukovic, 1961).

2.1.3.2 Age composition

Alosa alosa

The age structure of the spawning population ranges from 3 to 8 years old for both sexes, with the majority of females maturing at ages 5–6, and the males at ages 4–5 (Anon., 1979; Cassou-Leins and Cassou-Leins, 1981; Douchement, 1981; Eiras, 1981; Mennesson-Boisneau and Boisneau, 1990; Sabatié, 1993; Prouzet *et al.*, 1994; Martin-Vandembulcke, 1999; Véron, 1999; Lambert *et al.*, 2001; Mota *et al.*, 2015). The majority of the catches consisted of females aged 5–6 years and males aged 4–5 years. Similar findings were reported from the River Lima, Portugal (Alexandrino, 1996a). The observed mean age decreased with increasing latitude ranging from 5.57 to 4.01 years for males and 6.41 to 4.42 years for females; a similar pattern was evident for mean size (L_t) at maturity, which ranged from 500.7 to 388.5 mm for males and 558.9 to 435.3 mm for females (Lassalle *et al.*, 2008).

Alosa fallax fallax

In the more northerly populations (Severn, England), the spawning stock consists of females mainly aged 5–7 and males aged 4–6, with a maximum age of 12 and 10 for females and males respectively (Douchement, 1981; Aprahamian, 1982 and unpubl.; O'Maoileidigh, 1990; Sabatié, 1993; Véron, 1999; King and Roche, 2008; Nachón *et al.*, 2015a). Similarly, in the Curonian Lagoon at the mouth of the River Nyamunas (Lithuania), males are predominantly aged 5–7, with a range of 3–9 years, whereas females are predominantly aged 5–8, ranging from 5 to 10 years (Švagždys, 1999; Stankus, 2009). At the southern limit of their range (Sebou, Morocco), the spawning population

consists of fish aged 3–4 years old, with a maximum age of 6 and 5 years for females and males respectively (Sabatié, 1993).

The age structure of the Portuguese populations in the River Miño population (Spain/Portugal) is similar to that of the more southern stocks, with the exception of the River Tejo (Portugal), which appears younger than other populations (Alexandrino, 1996a; Nachón *et al.*, 2015a).

Alosa fallax rhodanensis

The age of the spawning population ranges from 3 to 8 years old for both sexes, with the majority being 5–6 years (females) and 4–5 years (males) (Douchement, 1981; Le Corre *et al.*, 1997). Although both sexes can mature as young as 2 years old (Le Corre *et al.*, 2000), the average age for the upstream migrants in the Rhône (France) was 4.16 years for males and 5.05 for females; whereas in the Aude (France), it was 4.7 years and 5.7 years for males and females, respectively (Le Corre *et al.*, 2000). In the Hérault (France) and Tavignano (Corsica), males ranged from 4 to 5 and 3 to 6 years and the females 4 to 6 and 5 to 6 years, respectively, although sample sizes were small (Le Corre *et al.*, 2000).

Alosa fallax nilotica

The age structure of the anadromous population in the River Neretva (Bosnia and Herzegovina) ranged from 2 to 10 years old, with the majority consisting of 5-year-old fish (Vukovic, 1961). In the River Po (Italy), the maximum age recorded was 7 years for both sexes (Serventi *et al.*, 1990).

2.1.3.3 Population dynamics

Alosa alosa

Until the end of the 20th century, the Gironde-Garonne-Dordogne (France) population was the largest allis shad population in Europe (Baglinière *et al.*, 2000) and was considered as a reference population (Martin-Vandembulcke, 1999). A dramatic drop in landings and in estuarine juvenile abundance (Rougier *et al.*, 2012) led to the Gironde basin's diadromous fish management committee implementing a total moratorium in 2008.

The relationship between stock and recruitment from the Gironde-Garonne-Dordogne (France) was found to fit a Ricker curve (Martin-Vandembulcke, 1999). Recent reanalysis by Rougier *et al.* (2012) identified that this relationship gave rise to a demographic Allee effect in the reproduction dynamics which, combined with high estuarine mortalities, could explain the population collapse. However, they were unable to prove the presence of density-dependant mechanisms necessarily associated with a demographic Allee effect. Based on these data, Lambert and Rougier (in prep) proposed a precautionary diagram (ICES, 2004) adapted from developments realized for the European eel population (ICES, 2010). Since the 1991 cohort, the population has never been in the orange and green zones (Figure 2.1) and entered the dark red zone (the depensatory trap) without escaping in 2002. With the hindsight knowledge of 2014, a massive reduction in anthropogenic mortalities should have been decided more than six years before the 2008 moratorium. Even with a very low fishing mortality in the estuary, the total anthropogenic mortality has increased during the last three years.

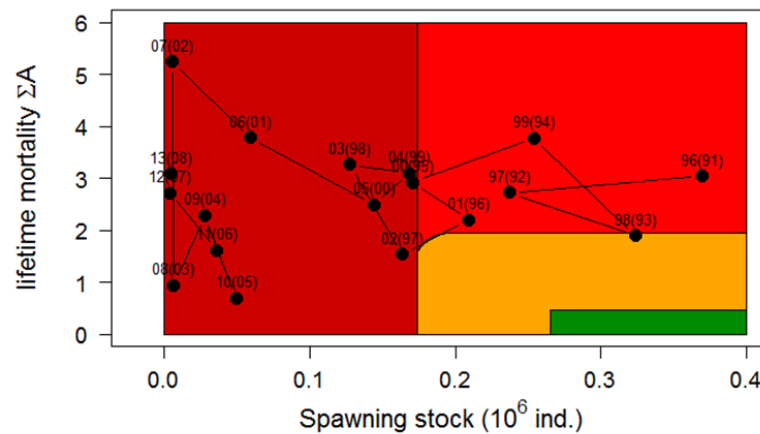


Figure 2.1. Precautionary diagram for allis shad (*Alosa alosa*) population in the Gironde system (two-digit labels indicate the years of anadromous migration and in brackets the year of birth).

In the Minho River, Portuguese official statistics confirmed mean catches of 200 t during the first half of the 20th century, with peaks of 300 t (Mota and Antunes, 2011). After the 1950s, catches decreased by ca. 90%, coinciding with the construction of the first dam on the river system (Baglinière *et al.*, 2003). In the last 40 years, mean annual catches reached about 4 t, with a peak in 1980 of about 18 t. However, these values are probably an underestimate owing to the lack of good official statistics. Unofficial Portuguese and Spanish data over the last eight years indicates that yields may have been twice as high (Mota *et al.*, 2015).

Fluctuations in abundance can partly reflect the influence of environmental factors. On the Loire River (France), Mennesson-Boisneau *et al.* (1999) found that recruitment of the 1980–1992 year classes was significantly correlated with flow during the period of upstream migration (15 March–15 June), though the relationship is heavily influenced by the flow in one year. The resultant implication is that in the Loire, the population is regulated by the amount of spawning and/or nursery area accessible. High flows allow the fish to penetrate further up the river system and increase the amount of rearing area available, reducing the level of density-dependent mortality. However, a more recent study of the 1995–2004 year classes found no relationship between juvenile (age 0+) abundance, and adult abundance, temperature, or flow (Boisneau *et al.*, 2008).

Alosa spp.

In the River Vilaine (France), the construction of the Arzal Dam in 1970 blocked access to the river. A fish pass was installed in 1996, and the number of *Alosa* spp. has increased steadily (Briand *et al.*, 2016). Presumably this increase initially occurred through straying from other populations, and, once established, the population expanded to fill the available habitat (Cédric Briand, pers. comm.).

Alosa fallax

Quantitative monthly sampling of fish and crustaceans at Hinkley Point 'B' Nuclear Power Station in the Bristol Channel (UK) has been carried out since October 1980 by P. Henderson (pers. comm.; Figure 2.2). The method is selective toward juvenile shad, with the majority of the catch consisting of the 0+ age group (Holmes and Henderson, 1990). Shad abundance shows high variability between years, making a status assessment very difficult.

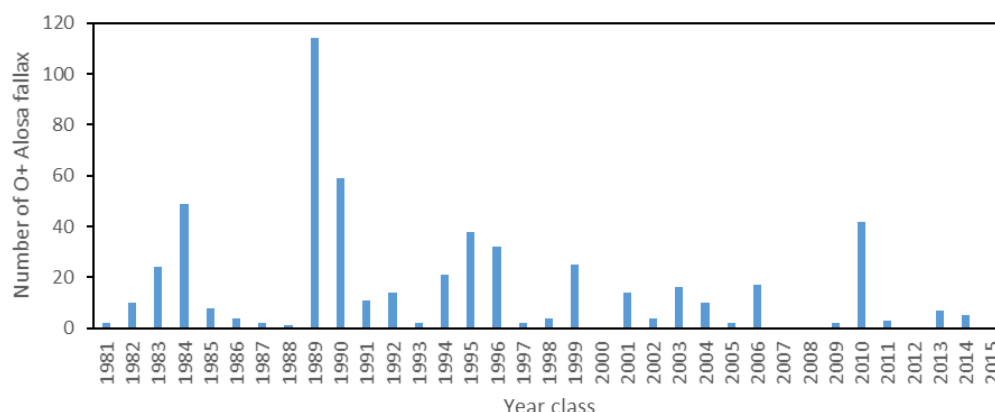


Figure 2.2. Monthly sampling of twaite shad (*Alosa fallax*) 0+ at Hinkley Point 'B' Nuclear Power Station in the Bristol Channel (UK). Source: P. Henderson (pers. comm.)

Part of the variation in recruitment can be associated with variation in temperature. For example, Aprahamian and Aprahamian (2001) found that mean July temperature explained the greatest proportion of the variance (67.1%) in year-class strength, followed by August (50.9%) and June (30.9%). Using the mean temperature for the three-month period improved the proportion of variability explained to 77.1%.

The relationship between stock (measured as the number of eggs deposited in a given year) and the number of recruits (measured as the number of eggs produced by females 6 years later, standardized using temperature as an explanatory variable) for the Severn (England) population of twaite shad, led to a weak density-dependent Ricker relationship (Aprahamian *et al.*, 2010).

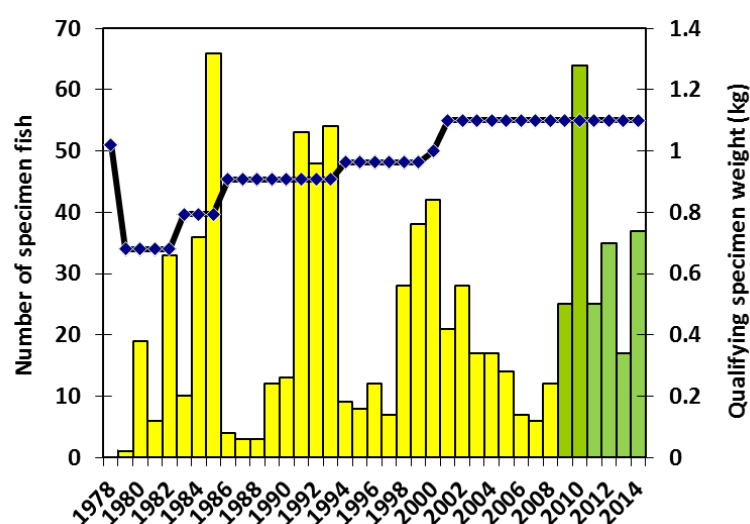


Figure 2.3. The spawning run of twaite shad (*Alosa fallax*), targeted for over 30 years by anglers fishing in the River Barrow estuary (Ireland) in May. Numbers represent rod-caught specimen of twaite shad ratified by ISFC 1978 and 2014. Solid line represents annual qualifying specimen weight. 2009 and 2010 includes confirmed shad hybrids. In early years, anglers were required to provide the body of the fish for confirmation (yellow). Since 2009, anglers are required to return all fish and to take a small sample of scales for genetic confirmation of species (green) (Lambert *et al.*, 2015). Source: www.irish-trophy-fish.com

Information on twaite shad status in Ireland has been largely derived from bycatch of commercial salmon netsmen operating in estuarine waters (King and Roche, 2008). An additional barometer has also come from data of the Irish Specimen Fish Committee

(ISFC) (www.irish-trophy-fish.com). This voluntary group sets “specimen” weights for a range of fish species of interest to anglers. If an angler catches a fish exceeding the “specimen” weight for that species, the angler receives a certificate. The spawning run of twaite shad has been targeted for over 30 years by anglers coming to the River Barrow (Ireland) estuary in May each year. The modal peaks (Figure 2.3) correspond to “angler effort” and may not solely reflect numbers of shad actually present. However, it is considered that the data indicate a fluctuating size for the spawning shad population. The anglers use social media to reflect the size of the run of fish – if the run is good, more anglers come and more fish are caught and released. If the run is poor, there is smaller angler effort. Where a large number of specimen fish were listed over a period of two or more years, the ISFC increased the “specimen” weight, making the challenge greater for the angler. Even with these increases, strong year classes are considered to be reflected in angler effort.

2.1.4 Genetics

2.1.4.1 Natal homing

Alexandrino (1996b) concluded that the genetic dissimilarity between geographically close populations of *Alosa fallax fallax* could only be explained if fish were homing to their natal river. Similar conclusions were reached using meristic data (Sabatié *et al.*, 2000). However, the same conclusion could not be reached for *Alosa alosa* because of the low level of polymorphism and population differentiation (Alexandrino *et al.*, 2006; Faria *et al.*, 2012). From the six anadromous populations analysed, only a slight divergence was observed in *Alosa alosa* from the southern and northernmost populations of the Guadiana (Spain/Portugal) and the Charente (France), respectively. Martin *et al.* (2015) observed for *A. alosa* that a significant proportion of individuals hatched and grew in a different watershed than the one in which they were collected. They concluded that although *A. alosa* exhibited a high fidelity to the natal river on an ecological time-scale, as inferred from otolith microchemistry, they showed weak genetic differentiation between collection sites. This suggests that the amount of straying is sufficient to explain the lack of a genetic structure for the Atlantic coast *A. alosa* populations.

Martin *et al.* (2015) and Randon *et al.* (2017) reported that although most *A. alosa* individuals returned to their natal watersheds, some fish did stray. This straying occurred most frequently between neighbouring river basins. Long-distance straying was evident, but not frequent, e.g. fish that were born in France, but were caught on their spawning migration in Portugal, or which were born in northwest France and caught in northern France. The low probability of long-distance straying between Portugal and northern populations was supported by the significant genetic differentiation of Portuguese and French populations (Alexandrino *et al.*, 2006) and the different body size of Portuguese spawners compared to other populations (Lassalle *et al.*, 2008).

2.1.4.2 Genetic population structure

Sabatino and Alexandrino (2012) analysed 21 microsatellite loci in 14 *A. alosa* and 23 *A. fallax* putative populations distributed across the present geographic area of distribution of both species.

Alosa alosa

Bayesian analysis of population structure (BAPS) identified 4 groups for anadromous *A. alosa* populations: 1 – the French populations from Charente, Garonne, Vienne, and the Solway in the UK; 2 – the Aulne in France; 3 – the Portuguese west populations from Minho, Lima, Mondego, and Tejo; and 4 – the southern Portuguese population

from Guadiana (Sabatino *et al.*, 2013). These results taken together with the results of Rougemont (2012), Jolly *et al.* (2012), and Martin *et al.* (2015) seem to define at least 5–6 distinct genetic groups in *A. alosa*: southern Portugal (Guadiana), west Portuguese populations (Minho, Lima, Mondego, Tejo), west French populations (Garonne, Dordogne, Charente, Loire), Brittany and southern UK populations (Aulne, Scorff, Tamar) and, eventually, a Normandy group (Vire, Orne) and a Biscay group (Nivelle) (Figure 2.4).



Figure 2.4. Genetically distinct anadromous *A. alosa* population groups/stocks based on clustering analyses with microsatellite loci data from Sabatino and Alexandrino (2012), Jolly *et al.* (2012) and Rougemont (2012).

Alosa fallax

Around 17 distinct groups were identified (13 corresponding to anadromous populations, Figure 2.5). Genetic differentiation among anadromous populations of *A. fallax* was consistently found throughout much of its range (Sabatino and Alexandrino, 2012). The following distinct genetic groups were defined (Sabatino *et al.*, 2013, 2015):

- Atlantic: 1 – Baltic Sea (Curonian Lagoon); 2 – North Sea (Nissum and Ringkøbing fjords, Denmark, Scheldt estuary, Belgium, Solway, UK); 3 – Severn group, UK (Severn, Wye, Usk); 4 – Towry, UK; 5 – western France (Charente); 6 – northwest Portugal (Minho, Lima, Mondego); 7 – southwest Portugal (Tejo, Mira); 8 – southern Portugal (Guadiana); 9 – Morocco (Sebou)
- Mediterranean: 10 – southern France (Rhône, Hérault, Aude); 11 – Corsica/Sardinia (Tavignano, Tirso); 12 – Adriatic (Po, Lake Skadar); 13 – Aegean Sea (Pinios, Izmir Bay)
- Landlocked populations: 14 – Killarney, Ireland; 15 – Lake Maggiore, Italy; 16 – Lake Como, Italy; 17 – Lake Garda, Italy.

Populations of *A. fallax* in almost all drainages were genetically isolated, including some less than a few hundred km apart. The pattern of isolation by distance observed in *A. fallax* indicates that this species appears to exhibit strong natal homing, as previously noted (Alexandrino, 1996a, 1996b; Alexandrino *et al.*, 2006; Jolly *et al.*, 2012).

For more information on the genetics of *A. alosa* and *A. fallax*, see Aprahamian *et al.* (2015).

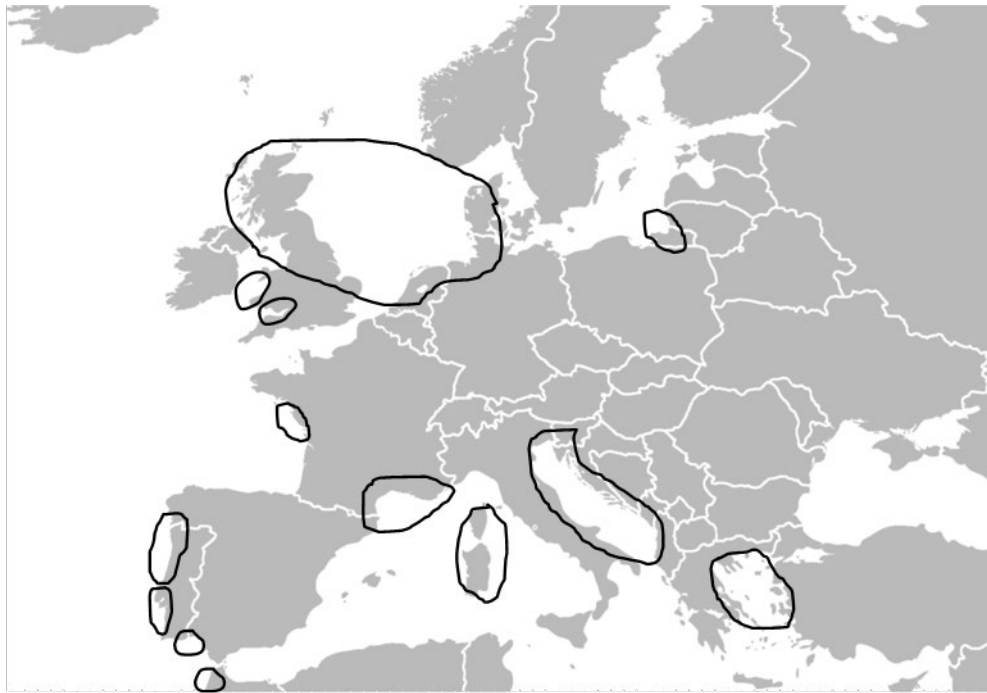


Figure 2.5. Genetically distinct anadromous *A. fallax* population groups/stocks based on clustering analyses of microsatellite loci data from Sabatino and Alexandrino (2012), Jolly *et al.* (2012) and Rougemont (2012).

2.1.5 International status with country highlights

Alosa alosa and *Alosa fallax* are both classified according to the IUCN Red List Least Concern Species category (Table 1.1; Freyhof and Kottelat, 2008a). In Italy, a recent revision of the regional IUCN Red List classified twaite shad as VU in Italian waters. This was the result of a prolonged revision process carried out by the national IUCN Validation Committee, where specific criteria for Italian populations were considered (Rondinini *et al.*, 2013).

Shads are listed in the EU Habitat Directive. Assessments made under the Habitat Directive are based on information on the status and trends of species populations or habitats, and on the main pressures and threats (figures 2.6 and 2.7).

It is important to note that for some countries there is not enough information about shads, making it difficult to evaluate the real conservation status. For example, in Spain, existing information is scarce and comes from general studies that only record presence or absence, like Doadrio *et al.* (2011).

The conservation status of *Alosa alosa* in the Atlantic region is “unfavourable–bad”, with declining populations. After the collapse of the Gironde allis shad population at the beginning of the 2000s (Rougier *et al.*, 2012), the Minho River population seems to be one of the largest populations in the southern part of the species’ distribution. Allis shad still has important commercial and heritage values in the Minho River (Mota and Antunes, 2011). There are very small numbers in Germany, Ireland, and The Netherlands; and landlocked populations exist in Portugal and Spain.

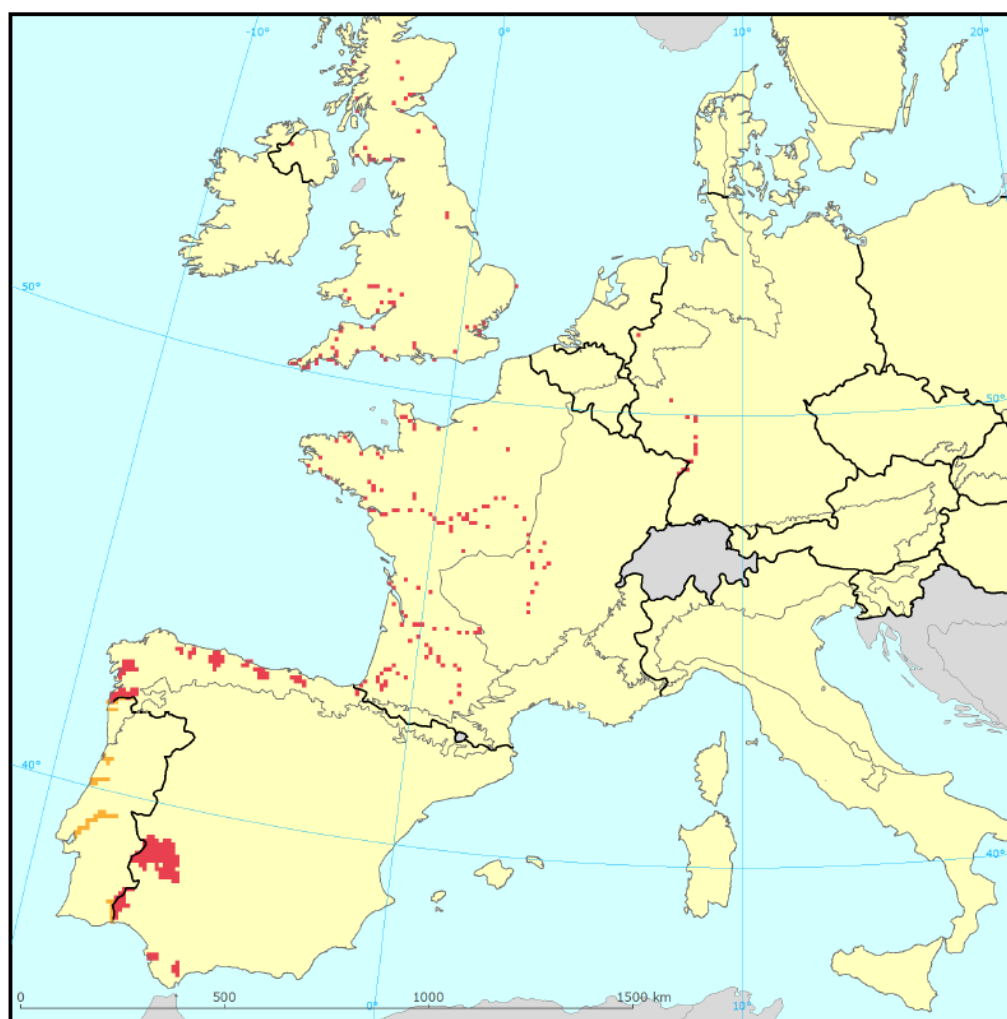


Figure 2.6. Conservation status of *Alosa alosa* at the European level for the 2007–2012 period, reported by member states (a version of this map is available at <http://bd.eionet.europa.eu/article17/reports2012/>). All populations have unfavourable–bad conservation status (red) with the exception of populations in Portugal (yellow; unfavourable–inadequate).

For *Alosa fallax*, the conservation status in the Atlantic region is “unfavourable–bad” and deteriorating. However, its status is not the same across the region, with cases of genuine improvement (Belgium and UK) and genuine deterioration (France). In Denmark, the species is considered marginal, with very few records in recent years. In Portugal and Spain its status is “unfavourable–inadequate”; whereas in France and Italy, there are improvements despite the “unfavourable–bad” status. In the Boreal region, its conservation status is “unknown”, owing to a lack of information in Latvia, where the population may not be self-sustainable according to the Article 17 report. However, its status is “favourable” in Lithuania, with stable populations.

2.1.6 Distribution summary

Alosa alosa and *Alosa fallax* have been reported from Iceland (Saemundsson, 1949; Valdimarsson *et al.*, 2012), at the northern-most limit of their range, to Morocco in the south (Furneston, 1952; Dollfus, 1955; Blanc *et al.*, 1976; Matallanas *et al.*, 1981) and as far east as Scandinavia (Pethon, 1979) and the Baltic Sea (Manyukas, 1989). Recently, *A. fallax* have been detected in the Azores and Madeira archipelagos (Barreiros *et al.*, 2014; Car-

neiro *et al.*, 2014). *Alosa alosa* have been reported only from the western part of the Mediterranean (Sostoa and Sostoa, 1979; Douchement, 1981), whereas *Alosa fallax* are encountered throughout the Mediterranean as far east as Turkey (Ceyhan *et al.*, 2012).

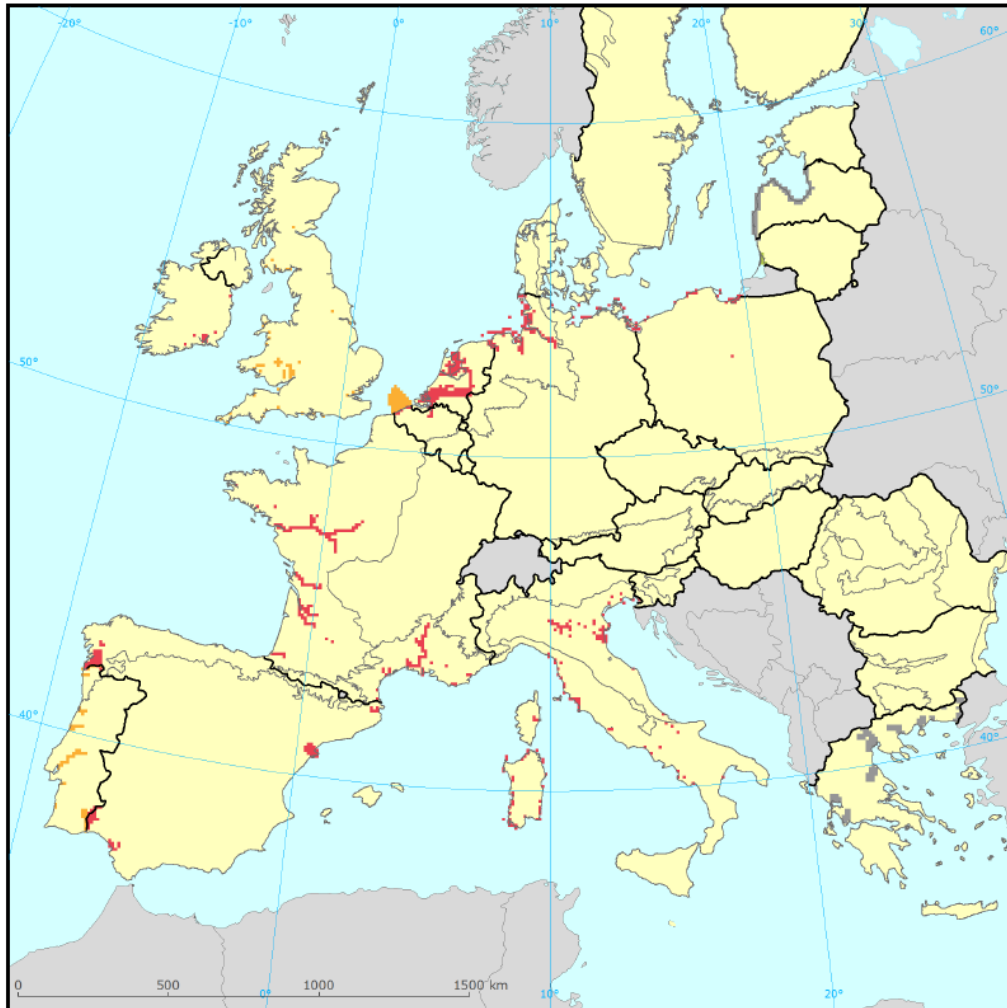


Figure 2.7. Conservation status of *Alosa fallax* at the European level for the 2007–2012 period, reported by member states (a version of this map is available at <http://bd.eionet.europa.eu/article17-reports2012/>). Most populations are yellow = unfavourable–inadequate (Portugal and UK) or red = unfavourable–bad (most populations).

2.1.7 Country-by-country changes in distribution

At present, it is estimated that there are 15 (possibly 17) self-sustaining populations of *Alosa alosa*, and 32 (possibly 37) populations of *Alosa fallax*, from countries that border the Baltic and Atlantic (Table A1.1 in Annex 1). In the Mediterranean, there are believed to be no self-sustaining populations of *A. alosa* and 44 (possibly 47) populations of *A. fallax* (Table A1.1 in Annex 1).

Populations of *Alosa alosa* have been lost from 12 rivers which drain into the Atlantic and three which drain into the Mediterranean (Table A2.1 in Annex 1). Six populations of *Alosa fallax* have been lost from rivers that drain into the Baltic, North Sea, and the Atlantic and seven populations from the Mediterranean (Table A1.2 in Annex 1). For both species, the main causes for the decline are dam construction, pollution, overexploitation, and habitat destruction. For the Rhine, it must be emphasized to note that the current project LIFE + “The reintroduction of the Allis shad (*Alosa alosa*) to the Rhine

system (LIFE06 NAT/D/000005⁶) is yielding satisfactory and positive results, with indications for some recovery of this population.

The presence of *A. alosa* in some small Cantabrian rivers has been reported by Doadrio *et al.* (2011), but virtually no other information exists to assess current status. It is possible that small populations of *A. alosa* or *A. fallax* persist or have become extirpated in small rivers, but lack of sufficient data means that it is not possible to evaluate status.

2.1.8 Threats

The main threats to both species are habitat loss and fragmentation, fishing both directed/targeted, and as bycatch, entrainment, pollution, and reduction in genetic diversity.

Table 2.1. The level of habitat loss caused by dams in some river systems which currently support or have previously supported self-sustaining populations of *Alosa* spp. (Nicola *et al.*, 1996; with additions, Mateus *et al.*, 2012; Nachón *et al.*, 2015b; Jorge Bochechas, pers. comm., <http://www.migrateurs-loire.fr/lineaire-accessible-pour-les-aloses/>).

River basin (Country)	Historic accessible length (km)	Current accessible length (km)	Percentage loss (relative to historical presence)
Severn (UK)	303	50	83.5
Rhine (DE)	1 320	986	25.3
Loire (FR)	1 024	753	26.5
Ulla (ES)	132	80	39
Minho (ES/PT)	340	80	76.5
Douro (PT)	207	21	89.9
Vouga (PT)	148	53	64.2
Mondego (PT)	258	75	70.9
Tagus (PT)	372	170	54.3
Guadiana (PT/ES)	834	130	84.4
Guadalquivir (ES)	560	106	81.1
Sebou (MA)	490	40	91.8
Ebro (ES)	928	117	87.4

2.1.8.1 Habitat loss and fragmentation

The construction of dams has been the primary factor affecting the abundance of populations of allis and, to a lesser extent, twaite shad over their entire distribution area (Table A1.2 in Annex 1; Aprahamian *et al.*, 2003; Baglinière *et al.*, 2003). Dams were initially often built without fish passes (fish ladders), or with fish passes that were either ill-adapted or did not accommodate the lack of jumping behavior in allis shad. The negative effect of dams depends on their number and position in the watercourse along the migratory route. The effects can be amplified by other factors, such as deterioration in water quality, and changes in the direction and shape of waterways. The level of habitat loss in some river systems that currently support, or have previously supported self-sustaining populations of *Alosa* spp. can be seen in Table 2.1.

⁶ http://ec.europa.eu/environment/life/project/Projects/index.cfm?fuseaction=search.dspPage&n_proj_id=3121

The impact of dams can be partly mitigated by providing a fishway or fish lift. However, the efficiency can be low. On the Dordogne, the estimated efficiency of three fishways situated over 30 km were: 55–65% at Bergerac (most downstream obstruction), 35–55% at Tuilieres, and 20–50% at Mauzac. At these levels of efficiency, < 5% of the shad that migrated upstream had passed the third dam at Mauzac (D. Courret and M. Chanseau, International Symposium on Restoration and Conservation of Shads, Bergerac, 14–15 October 2015)⁷.

A telemetry study by O. Guerri and P. Verdeyroux (International Symposium on Restoration and Conservation of Shads, Bergerac, 14–15 October 2015)⁷ further illustrates the problem of low passage efficiency for shad (Table 2.2).

Table 2.2. Fish pass efficiency at three passes on the Dordogne and Garonne (O. Guerri and P. Verdeyroux, International Symposium on Restoration and Conservation of Shads, Bergerac, 14–15 October 2015).

Number	Location / facility		
	Bergerac Double vertical slot	Tuiliers Single vertical slot	Golfech Fish lift
Tagged	27	74	121
Located at dam	4	16	4
Days present at dam	7	61	7
Active searches	4	18	7
Active searches with entry	2	0	6
Times fish entered the facility	3	0	15 ¹
Passing upstream	1	0	0

¹A fish can make more than one entry into the pass

The extraction of gravel has contributed to the decrease in stock abundance by damaging spawning sites and juvenile habitats. In the Adour River, the intensive extraction of beach gravel in a long stretch of spawning sites is linked to a decrease in commercial catches over 7 years from 36 t (1986) to 11 t (1993). In France this extraction has been recently reduced, and even halted in some rivers, by regulatory action (Taverny *et al.*, 2000).

2.1.8.2 Fishing (directed or targeted, and bycatch)

Fishing has rarely been the primary factor involved in the reduction of stock abundance in allis shad, although it has been invoked in the case of the Rhine population (de Groot, 1989). Further, normal stock assessments of allis shad have only just begun, and, until recently, there has not been sufficient data to analyse the impact of fishing (Rougier *et al.*, 2012). Nevertheless, it is clear that fishing pressure exerted on stocks that are challenged by other factors can enhance population decline. For example, in the Oued Sebou River, where the allis shad stock was already threatened by industrial pollution and dam building, maintenance of high fishing pressure over both time (e.g. no weekend ban) and space (sea, estuary, and spawning zones) seemed to force the

⁷ https://www.lanuv.nrw.de/alosa-alosa-2011/includes/docs/download/Zusammenfassung_Life_plus_Maifischsymposium_2015_en.pdf

stock to residual levels in many tributaries. A subsequent dam built near the estuary in the early 1990s brought the stock to extinction (Sabatié, 1993).

ICES Working Group on Bycatch of Protected Species (WGBYC) continues to compile and assess data from member state reports under Regulation 812/2004⁸ and/or from the DCF (ICES, 2014b). Clupeids are most commonly caught in bottom otter trawls. There appears to be some differentiation between the two species, with a higher preponderance of *Alosa alosa* being reported from set gillnets, whereas *Alosa fallax* appears to be more commonly caught in beam trawls and midwater pairtrawls.

Over the period 2009–2012, an average of ca. 4 t of *Alosa* sp. were reported as bycatch, with 3777 kg being declared from the North Sea and English Channel, Sweden, The Netherlands, and UK, and 258 kg (*Alosa fallax*) from Italian fisheries (ICES, 2014b). A recent study (Nachón *et al.*, 2015b) revised the official statistical data of marine landings (in kg) produced in all fish markets of Galicia (northwest of the Iberian Peninsula) over the 17-year period from 1997 to 2013. Up to 97.5% of *A. alosa* catches were landed at Rías Baixas, with A Guarda (7475 kg; 70.3% of the total catches) being the most important landing port. Cambados and Vigo accumulated 14.9% (1583 kg) and 6.4% (684 kg) of the total catches, respectively. The markets following in number of catches were Malpica (493 kg; 4.6%) and Corunha (259 kg; 2.4%), both located in Costa da Morte. All landings of *A. fallax* were made in Rías Baixas, with Baiona (215 kg; 96.4% of total catches) being the most important landing port. *Alosa alosa* catches showed notable interannual variations that were repeated cyclically throughout the entire time-series studied. Three peaks were recorded in 2000, 2004, and 2008, with values of ca. 1000 kg, after successive declines in catches. In the period from 1997 to 2007, there were no reports of *A. fallax* caught (with the exception of 2 kg caught in 2004). Since 2008, catches have occurred continuously, showing annual variations similar to those of *A. alosa*. The maximum catch occurred in 2009 (80 kg).

Taverny (1991b) investigated the potential impact of shrimp (*Palaemon longirostris*) and glass eel (*Anguilla anguilla*) fishery on the abundance of the 0+ population of shad in the Gironde Estuary. The glass eel fishery took less than 1% of both species. The impact of the shrimp fishery was only evaluated for *A. fallax*. 11–26% was estimated to be killed in the fishery.

2.1.8.3 Entrainment

Fish entrainment has been an issue in power station cooling water take-off situations (Claridge and Gardner, 1978; Aprahamian, 1988; Maes *et al.*, 2005). Taverny (1991b) investigated the potential impact on the abundance of the 0+ shad population of the power station in the Gironde Estuary (France), and showed that an estimated 2.5–5.8% and 9.6–11% of the *A. fallax* and *A. alosa* 0+ populations were affected, respectively.

2.1.8.4 Pollution

Barriers to migration can be created as a result of poor water quality, particularly of low levels of dissolved oxygen. In upper estuaries this can arise naturally, because re-suspension of the sediment is maximized, causing a sag in dissolved oxygen. This condition can be exacerbated by input of organic matter from domestic sewage and industry, causing a water quality barrier (Pomfret *et al.*, 1991). Such a barrier was perceived to exist in the Elbe Estuary (Germany). The improvement in water quality after 1991

⁸ <http://data.europa.eu/eli/reg/2004/812/oj>

(Gerkena and Thiel, 2001) was considered to be the main reason why *Alosa fallax* currently migrate an additional 20 km upstream to spawn (Costa *et al.*, 2002) compared to the situation in the 1960s (Hass, 1968) and 1980s (Möller and Dieckwisch, 1991).

The decline in the population of *A. fallax fallax* in the Nemunas (Neman/Nyamunas) River (Lithuania) has been attributed to the construction of the Kaunas hydroelectric dam combined with elevated pollution caused by poorly treated wastewater from paper mills in the lower river (Maksimov and Toliušis, 1999; Repečka, 1999, 2003a, 2003b; Žiliukas and Žiliukienė, 2002). During the 1970s and early 1990s, *A. fallax fallax* were only occasionally caught and were considered very rare in the Curonian Lagoon (Milėrienė, 1997; Repečka, 1999, 2012) as well as over the Baltic Sea region (Wiktor, 1989; Winkler, 1991; Thiel *et al.*, 2008). However, since the late 1990s, they have become increasingly more abundant (Repečka, 2003b, 2012; Maksimov, 2004), with large abundance of juveniles in the Baltic Sea and the northern part of the lagoon (Repečka, 2012). This may be the result of (i) a two- to three-fold reduction in phosphates, nitrates, and BOD7 in the River Nemunas and Curonian Lagoon, (ii) the deepening of the Klaipėda Strait (1984–1986), improving access to the Nyamunas (Repečka, 2003a, 2003b, 2005, 2012), and (iii) the ban on catching and landing of twaite shad (Maksimov, 2004).

Maes *et al.* (1998) and Vrielynck *et al.* (2003) reported that historically *Alosa fallax* used to spawn just above the tidal limit in the River Scheldt (UK) on sandy beaches near Schelle. The decline was associated with environmental degradation. However, twaite shad have recently been recorded from the lower river (Maes *et al.*, 2005, 2008), associated with an improvement in water quality.

The River Thames (UK) supported a spawning population of *Alosa fallax* up until the middle of the 19th century. Its decline has been attributed to a deterioration in water quality (Arahamian and Aprahamian, 1990).

2.1.8.5 Reduction in genetic diversity

The occurrence of hybrids may be indicative of a pressure preventing access to spawning, either in the form of low population levels, or of natural or man-made restrictions (Rameye *et al.*, 1976; Manyukas, 1989; Menesson-Boisneau *et al.*, 1993). The occurrence of hybrids of *A. alosa* × *A. fallax fallax* have been reported from Ireland (King and Roche, 2008; Coscia *et al.*, 2010), from the Solway Firth, UK (Maitland and Lyle, 2005; Jolly *et al.*, 2011), the Rhine, Germany (Hoek, 1888; Redeke, 1938), the French rivers Loire, Charente, Adour (Douchement, 1981; Boisneau *et al.*, 1992; Rougemont, 2012) and Aude (Douchement, 1981), from the River Ulla, Spain (Nachón *et al.*, 2015b), from the River Minho, Spain/Portugal (Mota *et al.*, 2015; Nachón *et al.*, 2015a, 2015b), from the Mondego and Lima, Portugal (Alexandrino, 1996a, 1996b; Alexandrino *et al.*, 2006), and from the Sebou, Morocco (Sabatié, 1993).

It has been suggested that the prevalence of hybridization is related to the presence of obstructions to the free passage of migrants upstream, resulting in the use of communal areas for spawning (Boisneau *et al.*, 1992), and there is some evidence of their temporal stability (Jolly *et al.*, 2011).

2.1.8.6 Main pressures and threats identified in the Habitats Directive

The 27 Habitats Directive member states (Report under Article 17 of the Habitats Directive Period 2007–2012) were asked to report the 20 most important threats and pressures using an agreed hierarchical list which can be found on the Article 17 Reference

Portal⁹. Pressures are activities which are currently having an impact on the species, and threats are activities which are expected to have an impact in the near future. Pressures and threats were ranked in three classes: high, medium, and low importance, with a maximum of 5 entries for high importance pressures or threats (Evans and Arvela, 2011). Table 2.3 only shows threats and pressures classed as high; for some species, there were less than 10 threats or pressures reported as highly important.

Table 2.3. Ten most frequently reported highly important pressures and threats for *Alosa alosa* and *Alosa fallax*, as identified by the 27 member states. See “Download factsheet” in Species Assessment at EU biogeographical level in Eionet (2012a, 2012b). A member state could identify up to five high importance pressures or threats; the numbers in the table represent the percentage of member states reporting that pressure or threat.

Activity	<i>Alosa alosa</i>		<i>Alosa fallax</i>	
	Pressure	Threat	Pressure	Threat
Changes in water body condition*	33	33	29	25
Other changes to ecosystems	24	29	19	20
Fishing and harvesting aquatic resources	24	19	17	18
Mining and quarrying	10	10	7	8
Pollution to surface waters			7	5
Invasive alien species			5	5
Use of pesticides in agriculture	5	5	2	5
Irrigation in agriculture	5	5	2	3
Shipping lanes and ports			2	3
Urbanization and human habitation			2	3

* Includes human induced changes in hydraulic conditions

For Atlantic populations of *Alosa alosa*, the main pressures were identified as overfishing, use of biocides, hormones and chemicals, obstacles to migration (dams), water abstraction, and dredging (Eionet, 2012a). In the Mediterranean region, the main pressures are pelagic trawling, canalization, loss of spawning habitat (dredging), water abstraction, and obstacles to migration such as dams (Eionet, 2012a).

For the Atlantic populations of *Alosa fallax*, the main pressures are fishing (including pelagic trawling), water pollution and abstraction, building of dams, inbreeding, and invasive alien species (Eionet, 2012b). In the Mediterranean region, the main pressures are fishing (including pelagic trawling), water pollution and abstraction, building of dams, inbreeding, and invasive alien species (Eionet, 2012b).

2.1.9 Conclusion – future

Like other diadromous fish, allis and twaite shad have suffered from the progressive impact of human activities in all the major river systems (Section 2.1.8). This has resulted in a pronounced fragmentation of their distribution area and a sharp decrease in their abundance levels. Some recent restoration programmes have started taking into account the biological specificity of these species. There are still a number of difficulties to be faced when restoring habitat connectivity: (i) lack of political and public awareness on the issue; (ii) lack of coordination among administrative organizations

⁹ http://bd.eionet.europa.eu/article17/reference_portal

across different parts of the river basins, and among river, estuarine, and marine jurisdictions; (iii) lack of knowledge of the species habitat requirements and the hydromorphology of each basin; and (iv) the low efficiency level of fishways (Mateus *et al.*, 2015). One of the issues recently highlighted is the impact of bycatch, and of fisheries as a whole (King and Roche, 2008; Trancart *et al.*, 2014; La Mesa *et al.*, 2015; Nachón *et al.*, 2015b). There may be a need to revise some of the legislation, in order to allow proper management and protection of both species. This should involve a review of the legislation, specifically focusing on fishing areas and the establishment of sanctuary areas, quotas, allowed methods, size limits, and closure periods. The problem of under, or false declaration of catches also needs to be addressed (Mateus *et al.*, 2015).

An increase in knowledge is necessary for the ecobiological aspects regarding stock-recruitment relationships and population dynamics through improved modelling (Aprahamian *et al.*, 2010; Rougier *et al.*, 2012). Furthermore, long-term data recording should be maintained in some index rivers and/or for certain populations to separate natural variations (global changes) from anthropogenic variations. This should include populations settling in newly colonized rivers. These data are crucial to forecasting fluctuations and establishing an objective basis for shad stock management, as well as for providing an integrated approach for natural resource management by catchment area.

At present, most of our understanding of the biology of *Alosa alosa* and *A. fallax* is based on the adult component of the population, and relatively little is known on the immature stages. Most studies have been carried out in estuaries and/or freshwater when the adults are concentrated on their spawning migration. Studies have been mainly carried out on those river basins draining into the Atlantic, whereas the Mediterranean populations remain poorly studied.

Some of the data presented in the section on stock structure are quite old. While this does not invalidate it, the data should be updated. The genetics of *Alosa alosa* and *A. fallax* is extensive and has been well researched (section 2.1.4). There are, however, virtually no studies undertaken in the marine environment, except for the investigations of Quero *et al.* (1989), Taverny (1991b), Sabatié (1993), Trancart *et al.* (2014), Nachón *et al.* (2015b), and La Mesa *et al.* (2015). This basically reflects sampling opportunities provided by the commercial fishery (often bycatch) and their economic importance. Similarly, though not presented here, our understanding of juvenile ecology is poor (Aprahamian *et al.*, 2015).

In the future:

- Conservation limits must be developed, particularly the limit for each river system below which a population should not fall (see Rougier *et al.*, 2012 and the section on population dynamics). To achieve this, there must be a commitment to obtain the required datasets for a better understanding of variations in life history and population parameters, population dynamics, and the relationship between stock and recruitment.
- Monitoring protocols need to be harmonized between countries, in order to permit comparisons. Failing this, methods must be intercalibrated among countries in the near future.
- Fisheries management and conservation management must be merged in a more comprehensive programme of monitoring (the Project MIGRANET is a good example of this; MIGRANET, 2012).

- The Habitats and Species Directive requires the assessment of the status of *Alosa* spp. against a predetermined set of conservation objectives, with the long-term goal of ensuring a favourable conservation status. The possibility should be assessed of using these species as a metric for habitat continuity or quality, and for assessing good ecological status. This could be achieved by setting up a transnational organization for the genus, possibly through ICES/EIFAAC, with terms of reference including stock assessments, development of biological reference points, and advice on monitoring and research of *Alosa* stocks and fishery .

2.1.10 Recommendations

- Improve our knowledge of Mediterranean stocks and update our knowledge of stock structure.
- Review current programmes for stock monitoring and conservation management, with focus in evaluating their suitability to achieve: (i) Good Ecological Status under the Water Framework Directive and (ii) Favourable Condition under the Habitats and Species Directive.
- Review fishery legislation.
- Improve public awareness of the status of the species.
- Improve our understanding of: (i) population dynamics and homing behaviour; (ii) habitat / ecological requirements – marine, estuarine, and freshwater; (iii) marine phase of the life cycle; and (iv) fish pass requirements.

2.2 European river lamprey

2.2.1 Taxonomy

Class: Petromyzontida
 Order: Petromyzontiformes
 Family: Petromyzontidae
 Scientific name: *Lampetra fluviatilis*

Subspecies, variations, and synonyms

Synonyms: *Petromyzon argenteus* Bloch 1795
Petromyzon branchialis Linnaeus 1758
Ammocoetes communis Gistel 1848

Petromyzon fluviatilis Linnaeus 1758
Petromyzon jurae MacCulloch 1819
Petromyzon omalii Beneden 1857
Lampetra opisthodon Gratzianov 1907
Petromyzon sanguisuga Lacépède 1800
 Eschmeyer *et al.*, 2015

Taxonomic notes

Lampetra fluviatilis adults have been separated by some authors based on size into “typical” and “praecox” forms. The term “praecox” refers to smaller forms that spend less time feeding in the sea.

Common names

EN: European river lamprey; PT: Lampreia-de-rio; ES: Lamprea de río; FR: Lamproie de rivière; IT: Lampreda di fiume; NL: rivierprik; DE: Flussneunauge; PL: Minóg rzeczny; LT: Upinė nėgė; LV: Upes nēģis; ET: Jõesilm; FI: nahkiainen; SV: Flodnejonöga; GA: Loimpre abhann; RU: Evropeiskaya rechnaya minoga

General characteristics

Length: 18.5–49.2 cm
 Weight: 0.03–0.15 kg (*forma typica*; upstream migrants)
 Maximum age: 7 years
 Generation length: 7 years (varies among locations and depends on environmental conditions; Potter, 1980; Hardisty, 1986a)

The European river lamprey is an anadromous species distributed along the Baltic and North Sea coasts, and the Atlantic seaboard extending to the western Mediterranean. Landlocked populations exist in a number of lakes (Hardisty, 1986a; Inger *et al.*, 2010).

Even though globally classified as LC, the population status of *Lampetra fluviatilis* varies greatly among countries. There are serious conservation concerns in the southern distribution (e.g. it is considered CR in Portugal; Cabral *et al.*, 2005), but populations are more stable in central and northern countries, where the species is exploited commercially either for use as bait (e.g. England; Masters *et al.*, 2006) or for human consumption (e.g. Finland, Latvia, Estonia, and Sweden; Saat *et al.*, 2003; Sjöberg, 2011).

2.2.2 Life cycle and migrations

The life cycle of the anadromous river lamprey can be divided into two distinct phases: (i) a freshwater larval phase, and (ii) an adult marine phase. For periods of several years, the ammocoetes (lamprey larvae) lie burrowed in fine sediment deposits of rivers and streams, filtering from the water the micro-organisms and organic particles on which they feed (Hardisty and Potter, 1971a). The larva then undergo metamorphosis, which is characterized by the development of the oral disk, the appearance of teeth, eruption of the eyes, enlargement of the fins, and changes in pigmentation (Hardisty and Potter, 1971b; Manzoni *et al.*, 2015). Internal changes include major modifications to the digestive system, such as formation of a new esophagus, a remodelled intestine, and loss of the hepatic biliary tree and the gall bladder. Several changes are also observed in the respiratory and skeletal systems (Manzoni *et al.*, 2015). The metamorphosis precedes the downstream trophic migration and the onset of feeding, when the animal may be regarded as a young adult. The main downstream migration of *L. fluviatilis* occurs from late winter to early spring (Hardisty *et al.*, 1970; Potter and Huggins, 1973). The duration of the marine phase, specificity of hosts, distances travelled, and volitional control over journeys are areas of limited knowledge. The marine trophic phase occurs over an approximately 2-year period, during which they feed primarily on blood and muscle tissue of fish.

Following the completion of their marine trophic phase, adult anadromous lampreys re-enter freshwater and migrate to upstream river stretches where they build nests, spawn, and die shortly thereafter (Hardisty, 1986a; Moser *et al.*, 2015). Anadromous lamprey at sea are not considered to have a homing instinct to guide them to the basin where they were born. Rather, it is thought that they are drawn into particular catchments by pheromones released by the ammocoetes or larvae already residing there (Moser *et al.*, 2015). The timing of the river lamprey's upstream migration varies among regions, but they typically migrate upriver in autumn/winter and spawn in spring (Hardisty, 1986a; Johnson *et al.*, 2015; Moser *et al.*, 2015). In some regions, there are two peaks of migration, one in autumn (when the fishing pressure is higher), and the other shortly before spawning (Saat *et al.*, 2003).

2.2.3 Stock structure and population dynamics

River lamprey fisheries exist in France, UK, Sweden, Finland, Lithuania, Estonia, Latvia, Poland, and the Russian Federation (Saat *et al.*, 2003; Renaud, 2011; Sjöberg, 2011). Former fisheries in Germany have apparently ceased. Declining stocks were noted as early as 1980. Tuunainen *et al.* (1980) stated that "...formerly, most Finnish rivers flowing to the Baltic Sea supported river lamprey stocks, but because of changes in the natural state of the rivers (e.g. because of dams, pollution, etc.) many stocks have been destroyed or weakened". At present, river lamprey fishing for food consumption is mainly concentrated in the Baltic Sea area (Sjöberg, 2011). An outline of some of the principal current and former fisheries is given below

Estonia

Lamprey fishing is important in Estonia. During the spawning run, lamprey ascend most Estonian rivers that run into the Baltic Sea (Saat *et al.*, 2003). Catches have decreased over the last 60–70 years as a result of damming, and fluctuate considerably among years. Average annual catches of 67 t were recorded between 1928 and 1938, followed by a decrease to mean values of 21 t between 1959 and 1968, and a subsequent recovery back to 67 t per year between 1994 and 2015 (Saat *et al.*, 2003; Sjöberg, 2011; www.agri.ee). There is a closed season for lamprey fishing in Estonia.

Finland

Statistics on river lamprey catches show total catches of ca. 1.8–3.0 million lamprey per year in the 1970s and 1980s and ca. 0.6–1.8 million lamprey per year in 2000–2011 (reviewed in Maitland *et al.*, 2015).

Latvia

Lamprey harvest is significant in Latvia, with the biggest annual catches registered in the late 1960s and early 1970s (410 t in 1971). During the 1970s, the size of the annual lamprey catch dropped sharply, with only 8 t harvested in 1980. However, during the 1980s, the annual lamprey catch began to increase, and currently river lamprey resources have recovered from their decline and are stable, though the annual catch rarely exceeds 100 t. From 2000 to 2012, the total annual catch fluctuated from 72.1 t (2010) to 136.4 t (2000). Lamprey fishing effort and catch size has remained stable in the last few decades. Lamprey fishing is commercial, and is strictly regulated (Abersons and Birzaks, 2014), with a closed season for lamprey fishing. The general impression among fishers in Latvia is that the demand for lampreys is higher than what they can deliver, and the prices for lampreys are at the higher end of the scale for fish (Sjöberg, 2011).

Poland

Annual catches in the Vistula River exceeded 100 t in 1930–1939, but by the end of the 1950s, catches were so low that fishing ceased. This was probably as a consequence of environmental pollution, overfishing, and river damming (Thiel *et al.*, 2009).

Sweden

Lamprey fishing occurred in 25 rivers shortly after 1950, compared to only 14 by the early 2000s. The construction of hydroelectric power plants in many of the main rivers has led to a strong decrease in the number of lampreys harvested, from ca. 200 000 per year in 1942–1951 in just one of the rivers fished for this species (Sjöberg, 2011), to ca. 150 000 in 2010–2011 for all rivers. There are no general rules regulating lamprey fishing in Sweden.

UK

The European river lamprey was formerly fished extensively in several rivers in Britain. In the main English lamprey fishery located in the River Ouse, total catches ranged from 25 500 to 54 500 lamprey per year between 1908 and 1914. In recent years, river lampreys have been caught in the River Ouse and sold to anglers for use as bait. It was estimated that 9083 to 30 992 lamprey were caught annually between 2000 and 2004 in this river (Masters *et al.*, 2006). Although annual catches in the River Ouse lamprey fishery have varied widely since 1995, catch per unit of effort did not decline between 2000 and 2012 (Foulds and Lucas, 2014). The relative exploitation level of the commercial fishery on the tidal Ouse was estimated to be between 9.9 and 12.0% in 2003–2004 (Masters *et al.*, 2006). Lampreys were originally considered bycatch in a licensed eel fishery, but lamprey catches were so high that it was essentially a commercial lamprey fishery. This led to the introduction of lamprey-specific legislation in 2011, including a cap on fishing licenses, catch quotas, and restricted fishing seasons (Foulds and Lucas, 2014). However, given the lack of reliable data on demographic processes, and the difficulties experienced in establishing an accurate exploitation rate, it is not yet known what would constitute a sustainable catch level in this river (reviewed in Maitland *et al.*, 2015).

2.2.4 Genetics

Until the advent of molecular markers as a common tool to improve understanding of systematics, the taxonomic validity of some members of this species was questioned. Specifically, some authors referred to members of paired lamprey species as morphs (alternative life-history strategies) of a single species (e.g. Enequist, 1937). Lamprey “paired species” consist of closely related lampreys, indistinguishable as larvae, but with distinct life histories as adults: one is parasitic and anadromous, and the other is non-parasitic and resides in freshwater. The non-parasitic form derives from a form similar to that of an extant parasitic lamprey (Hubbs, 1925, 1940; Zanandrea, 1959). Some parasitic ancestors have given rise to two or more different non-parasitic derivatives, and these are called “satellite species” (Vladykov and Kott, 1979).

High-resolution genetic data, namely restriction site-associated DNA sequencing (RADseq), were used to unveil the long-standing question about the taxonomic validity of the European brook (*L. planeri*) and river lampreys, revealing that they are two valid species that diverged recently (Mateus *et al.*, 2013). In this study, performed on lamprey populations from Portugal, most of the genes identified as showing fixed allelic differences between the two species had been previously related to functions implicated in the adaptation to a migratory vs. resident lifestyle in lampreys and bony fish (Mateus *et al.*, 2013).

Recent investigations on the paired European river and brook lampreys, using both mtDNA sequence data and polymorphic microsatellite DNA loci, revealed different scenarios for genetic differentiation and population structure between the two species in different parts of their distribution (Espanhol *et al.*, 2007; Bracken *et al.*, 2015; Rougemont *et al.*, 2015; Mateus *et al.*, 2016). This was related to post-glacial colonization patterns and the timing of the establishment of each population (Mateus *et al.*, 2016). The current population structure is strongly related to the ice ages. There is evidence that the previously glaciated areas in northern Europe may have been colonized by migratory lampreys (*L. fluviatilis*-type) expanding out of Iberian refugia (Mateus *et al.*, 2016). The general pattern observed is that gene flow is high among populations of river and brook lampreys in central/northern regions, but low between central/northern populations and the southern populations. In the southern glacial refugia, populations show high levels of differentiation, reflecting long periods of isolation as a result of glaciation. On the other hand, as a result of their more recent common ancestor, populations from northern Europe are less divergent, and there is evidence of strong recent gene flow among populations (Mateus *et al.*, 2016).

In general, anadromous populations are less divergent, as a result of their higher mobility and less restricted gene flow. However, the southern *L. fluviatilis* from Portugal revealed a genetic signature similar to a resident species, an almost absent ongoing gene flow with other *L. fluviatilis* populations, and a high degree of isolation and differentiation. The authors suggested that this population resembles the smaller “praecox” form, with limited dispersion movements, and is apparently isolated from the remaining European populations (Mateus *et al.*, 2016). The “praecox” forms have mean lengths of ca. 24 cm and mean weights of about 22 g (Abou-Seedo and Potter, 1979). Specimens from the Tagus River basin analysed by the authors had, on average, 26 cm total length and 33 g weight, though the smallest individual was 20 cm total length and 19 g weight (Mateus *et al.*, 2016). These low values contrast, for example, with those registered by Kemp *et al.* (2010) for this species in northeast England of 36.3 cm and 80.7 g.

The pair *L. fluviatilis*/*L. planeri* is apparently at different stages of speciation in different locations. There is evidence for high reproductive isolation in the southern refugium and low differentiation in the north, resulting from differences in the timing of colonization and refugial persistence (Mateus *et al.*, 2016). This speciation continuum occurs not only at the larger European north/south, but also at smaller scales, such as within countries. For example, in France, Rougemont *et al.* (2015) found variable levels of gene flow between sympatric populations of *L. planeri* and *L. fluviatilis*, ranging from panmixia to moderate differentiation, indicating a gradient of divergence. In the British Isles, microsatellite markers revealed strong differentiation among freshwater-resident *L. planeri* populations, and between *L. fluviatilis* and *L. planeri* in most cases, but little structure was evident between anadromous *L. fluviatilis* populations (Bracken *et al.*, 2015).

2.2.5 International status with country highlights

International

OSPAR Convention:	Not listed
EC Habitats Directive:	Annex II and V
CITES:	Not listed
Bern Convention:	Appendix III (2002)
Bonn Convention:	Not listed
IUCN Criteria:	–
Global IUCN Red List Category:	LC (Freyhof, 2013; Freyhof and Brooks, 2011)
European IUCN Red List Category:	LC (Freyhof, 2013; Freyhof and Brooks, 2011)
HELCOM Red List:	NT (HELCOM, 2013a)
	Previous classification: EN (HELCOM 2007)

Protection and Red List status by country

Belgium (Flanders):	– / NT (Verreycken <i>et al.</i> , 2014)
Belgium (Wallonia):	– / RE (Philippart, 2007; Kestemont, 2010)
Czech Republic:	– / RE (Lusk <i>et al.</i> , 2004)
Denmark:	– / DD (HELCOM, 2013a)
Estonia:	Fishery regulations / LC (HELCOM, 2013a)
Finland:	In some closed rivers, stocks are kept alive by transferring individuals over dams and in some others by stocking / NT (HELCOM, 2013a)
France:	49 Natura 2000 sites were designated for this species / VU (IUCN France <i>et al.</i> , 2010)
Germany:	Protected by national and European law (Annex II and V, Habitat Directive) / CR (Baltic Sea) (HELCOM, 2013a)

Ireland:	SACs were listed on the Irish east coast estuaries and in the large Shannon estuary; Fisheries Acts 1959–2006; Fisheries Act (Northern Ireland) 1966; Foyle Fisheries Act (NI) 1952; Foyle and Carlingford Fisheries Act 2007 / LC (King <i>et al.</i> , 2011)
Italy:	– / CR (Rondinini <i>et al.</i> , 2013)
Latvia:	Fishery regulated through the number of allowed gears and a closed season. Under the Law on the Conservation of Species and Biotopes Nr. 396 and 45 / – (HELCOM, 2013a)
Lithuania:	– / – (HELCOM, 2013a)
Norway:	– / LC (Kålås <i>et al.</i> , 2010)
Poland:	Prohibited to kill, catch, or disturb this species under strict protection / VU (HELCOM, 2013a)
Portugal:	One SCI under the Habitats Directive: Estuário do Tejo (PTCON0009). However, the site is restricted to the river estuary and only coincides marginally with the real geographic distribution of <i>L. fluviatilis</i> . Ten locations have been selected to be proposed as SACs for lampreys in the Tagus basin (Ferreira <i>et al.</i> , 2013), of which eight presumably support populations of <i>L. fluviatilis</i> , as no obstacle to the migration is known to occur / CR (Cabral <i>et al.</i> , 2005)
Russian Federation:	Excluded from the Red Book of the Russian Federation since 1997 (HELCOM, 2013a)
Spain:	– / RE (Doadrio, 2001)
Sweden:	Regional restoration programmes in rivers, and a national action programme is currently under development / LC (ArtDatabanken, 2015)
UK:	Nine SAC rivers in England and Wales; six SAC rivers in Scotland, including the River Endrick, which is the major spawning stream for a land-locked population of river lamprey present in Loch Lomond / VU (Maitland, 2000)

2.2.6 Distribution summary

Lampetra fluviatilis is restricted to European watersheds. Its range extends from southern Norway (around Bergen), along the Baltic and North Sea coasts, the Atlantic waters of Britain and Ireland, France, and Portugal (River Tagus), and on to the western Mediterranean (along French and western Italian coasts; Hardisty, 1986a) (Figure 2.8). It has also been reported in Turkey (Erguven, 1989). The river lamprey is generally a common and widely distributed member of the ichthyofauna of the Baltic Sea (Thiel *et al.*, 2009). The species has been occasionally recorded in the Adriatic and Ionian seas, but they are absent from the Black, Caspian, and Polar seas. Landlocked populations are known from lakes Ladoga and Onega, and the Volga basin (The Russian Federation), Loch Lomond (Scotland, UK), some lakes in Finland, and Lough Neagh (Northern Ireland, UK) (Goodwin *et al.*, 2006).



Figure 2.8. Distribution of European river lamprey. Yellow shows extant range; red indicates extinct populations. (IUCN Version 2015-4).

Table 2.4. Historical and present distribution of *L. fluviatilis* by country, indicating presence or absence in current or historical distribution rivers.

Country	River	Distribution		Trend direction
		historic	present	
Portugal	Minho	X ¹		
	Mondego	X ¹		
	Tagus	X ¹	X ²	Unknown
France	Hem	?	X ³	
	Aa	?	X ³	
	Bresle	?	X ³	
	Béthune	?	X ³	
	Risle	?	X ³	
	Odon	?	X ³	
	Oir	?	X ³	
	Loire	?	X ³	
	Dordogne	?	X ³	
	Garonne	?	X ³	
Estonia	Most coastal rivers	X ⁴	X ^{4,5}	Decreased, stable
Sweden	Most coastal rivers	X ⁶	X ⁶	Decreased, stable
UK	Nidd	?	X ⁷	
	Swale	?	X ⁷	
	Ure	?	X ⁷	
	Derwent	?	X ⁷	
	Trent	?	X ⁷	
	Wear	?	X ⁷	
	Dee	?	X ⁷	
	Endrick Water (Loch Lomond)	?	X ^{7*}	

Table 2.4. (continued)

Country	River	historic	present	Trend direction
Ireland	Boyne	?	X ⁸	Unknown
	Liffey	1889, 1899 ⁹	X ⁸	Unknown
	Avoca	X ¹⁰	X ⁸	Unknown
	Owenavarragh		X ⁸	Unknown
	Slaney	X ¹⁰	X ⁸	Unknown
	Barrow	X ¹⁰	X ⁸	Unknown
	Nore		X ⁸	Unknown
	Suir	X ¹⁰	X ⁸	Unknown
	Munster Blackwater	X ¹⁰		
	Laune	X ¹⁰		
	Mulkear		X ⁸	Unknown
	Maigue		X ⁸	Unknown
	Shannon tidal tributaries	X ¹⁰	X ^{8, 11}	Unknown
	Garavogue (Lough Gill)	¹⁰		
Poland	Vistula basin	X ¹²	X ^{12, 13}	Unknown
	Oder basin	X ¹²	X ^{12, 13}	Unknown
	Polish coastal rivers	X ¹²	X ^{12, 13}	Unknown

¹ Baldaque da Silva (1891); ² Mateus *et al.* (2012, 2016); ³ Rougemont *et al.* (2015); ⁴ Saat *et al.* (2003); ⁵ www.agri.ee; ⁶ www.artfakta.se; ⁷ Bracken *et al.* (2015); ⁸ NPWS (2013); ⁹ National Museum of Ireland; ¹⁰ Kurz and Costello (1999); ¹¹ Igoe *et al.* (2004); ¹² Witkowski (2010); ¹³ Fishery data; inquiries conducted by NMFRI employees

* Anadromous and landlocked

2.2.7 Country-by-country changes in distribution and data

The European river lamprey is known to have disappeared from a number of European rivers, and the general abundance of the species has declined (tables 2.4 and 2.5). Although river lamprey catches have declined, exploitation continues to be high at a local level, and overexploitation may represent a significant ongoing threat in some regions (Maitland *et al.*, 2015). At present, the tradition of river lamprey fishing for food consumption is concentrated mainly in the Baltic Sea area. However, reports indicate remarkable decreases in catches during the last decades in this area (Sjöberg, 2011).

Finland

This species previously had a widespread distribution throughout Finland in all accessible river systems. However, after extensive dam construction in Finnish watercourses, the distribution has declined dramatically, especially over the past century. There are, however, still a number of rivers with high annual catches, and the importance of the fishery is high in certain regions (ICES, 2005).

Germany

River lamprey populations have decreased since the mid-1950s. In former times, there was an important river lamprey fishery in German rivers emptying into the southern Baltic Sea. However, this seems to have since ceased (Sjöberg, 2011).

Table 2.5. Conservation status by country, with a comparison between the status reported for river lamprey by the Study Group on the Status of Diadromous Fish Species (SGSDFS) in 2005 (ICES, 2005) and the current situation. In 2005, the group identified four countries holding threatened populations of river lamprey (France: VU; Germany: EN; Italy: EN; and Poland: VU). Of these, France and Poland retained the same classification, but in Germany and Italy, the river lamprey is now classified as Critically Endangered (CR). In addition, this species passed to being classified as VU in UK and as CR in Portugal (in Portugal, this classification was recognized in 2005 (Cabral *et al.*, 2005), but was not included in the report.

National classification		
Country	2005	Last assessment
Belgium (Flanders)	NT	NT (Verreycken <i>et al.</i> , 2014)
Denmark	Indeterminate	DD (HELCOM, 2013a)
Finland	No category assigned	NT (HELCOM, 2013a)
France	VU	VU (IUCN France <i>et al.</i> , 2010)
Germany	EN	CR (Baltic Sea) (HELCOM, 2013a)
Ireland	Indeterminate	LC (King <i>et al.</i> , 2011)
Italy	EN	CR (Rondinini <i>et al.</i> , 2013)
Norway	No category assigned	LC (Kålås <i>et al.</i> , 2010)
Poland	VU	VU (HELCOM, 2013a)
Portugal	No category assigned	CR (Cabral <i>et al.</i> , 2005)
Spain	EX	RE (Doadrio, 2001)
Sweden	NT	LC (ArtDatabanken, 2015)
UK	No category assigned	VU (Maitland, 2000)

DD: data deficient; VU: Vulnerable; NT: Near Threatened; EN: endangered; CR: Critically endangered; RE: regionally extinct; EX: extinct; NA: not applicable

Ireland

In reporting to the EU, under Article 17 of Habitats Directive, Ireland uses larval lamprey information, and this leads to a combining of brook and river lamprey data (Table 2.6). Discrete information on river lamprey is confined to adults, who are reported from a number of rivers along the east and southeast coasts and from some rivers in the estuary of the River Shannon. The freshwater migration of adults appears to be extended over the autumn–winter–spring period prior to spawning, and repeat use of particular spawning sites has been observed. Autumn-caught adults on the River Slaney measured 28–36 cm (King and Linnane, 2004), with a pronounced reduction in length range in spring samples, as also recorded for Polish rivers (Witkowski and Koszewski, 1995).

Poland

River lamprey used to be very common, but it is now less numerous, and may have disappeared in some rivers (ICES, 2005). Due to Polish law regulation, with strict protection of the species, and fishery regulation (e.g. high selectivity fish traps), it is not possible to obtain relevant data concerning changes in distribution and abundance using commercial fishery methods.

Portugal

River lampreys were historically reported in three river basins (Baldaque da Silva, 1891) but are now confined to the lower stretches of one river basin. In the information

provided in the national report under Article 17 of the Habitats Directive by the Portuguese Environmental Agency, covering the period 2007–2012, *L. fluviatilis* was globally classified as “U2” (unfavourable-bad stable); population evaluation was “XX” (unknown), habitat and range evaluation was “U1” (Unfavourable inadequate), and future perspectives was “U2” (unfavourable bad)¹⁰.

Table 2.6. Monitoring programmes and data for river lamprey.

Country	Monitoring programme	Available data
Estonia	Monitored in conjunction with the general survey of protected fish species in Estonia	Survey and official landings data from the rivers
Ireland	Programme to establish anadromy and spawning locations – one catchment/year	http://www.fisheriesireland.ie/research/habitats/541-habitats-directive-and-red-data-book-fish-species-summary-report-2014
Poland	---	Piecemeal data from Vistula and Szczecin Lagoon as a part of monitoring conducted within Multiannual Programme for Collection of Fisheries Data.
Sweden	Electrofishing for monitoring (larvae)	1988–present; >200 rivers and streams

2.2.8 Threats

The main threats identified for European river lamprey are the construction of dams and weirs in rivers, which block upstream migration to the spawning grounds, and aquatic pollution. Occasional severe droughts may also be a threat, as well as peaking mode in hydropower plants. Overfishing and poaching may also be affecting this species in some areas where it is still commercially important (either for food consumption or as bait).

2.2.9 Conclusion - future

The progressive development of fishing gear and an increasing intensity of fishing effort has resulted in a considerably greater pressure on river lamprey populations. The effect of fishing pressure has been compounded by habitat degradation, such as pollution and river damming.

Management recommendations:

- Habitat restoration: (i) barriers to migration should be equipped with suitable fish pass devices efficient for lampreys; (ii) water quality should be monitored and improved.
- Maintain sustainable fishery and stocks: legislation should be applied in all countries holding river lamprey fisheries, applying, for example, a cap on fishing licenses, catch quotas, and restricted fishing seasons.

¹⁰ <http://www2.icnf.pt/portal/pn/biodiversidade/rn2000/dir-ave-habit/rel-nac/rel-nac-07-12>

2.3 Sea lamprey

2.3.1 Taxonomy

Class:	Petromyzontida
Order:	Petromyzontiformes
Family:	Petromyzontidae
Scientific name:	<i>Petromyzon marinus</i> Linnaeus, 1758

Subspecies, variations, synonyms

Eschmeyer <i>et al.</i> 2015	<i>Petromyzon americanus</i> Lesueur 1818
	<i>Petromyzon (Bathymyzon) bairdii</i> Gill 1883
	<i>Ammocoetes bicolor</i> Lesueur 1818
	<i>Petromyzon marinus dorsatus</i> Wilder in Jordan and Gilbert 1883
	<i>Petromyzon lampetra</i> Pallas 1814
	<i>Petromyzon maculosus</i> Gronow in Gray 1854
	<i>Petromyzon nigricans</i> Lesueur 1818
	<i>Petromyzon ruber</i> Lacépède 1800
	<i>Petromyzon marinus unicolor</i> Gage 1928
	<i>Oceanomyzon wilsoni</i> Fowler 1908

Common names

EN: Sea lamprey; PT: Lampreia marinha; ES: Lamprea marina; FR: Lamproie marine; IT: Lampreda di mare; GA: Loimpre mhara; DE: Meerneunauge; DA: Havlampret; LT: Jūrinė nėgė; LV: Jūras nēģis; SV: Havsnejonöga; PL: Minóg morski; FI: Merinahkiainen; RU: Morskaja minoga; NL: Zeeprik; ET: Merisutt.

General characteristics

Length:	11.4–120.0 cm total length (TL)
Body weight:	2.3 kg (adult) for a 120.0 cm TL individual (Hardisty, 1986b)
Duration of larval life:	3–7 years (e.g. Quintella <i>et al.</i> , 2003; Dawson <i>et al.</i> , 2015; Silva <i>et al.</i> , 2016a)
Duration of postmetamorphic life in anadromous populations:	Ca. 2 years (Renaud, 2011; Silva <i>et al.</i> , 2013a)

The sea lamprey is an anadromous species distributed on both sides of the North Atlantic. Several landlocked populations inhabit the North American Great Lakes (Laurentian Great Lakes, Finger Lakes (Lake Cayuga), Lake Champlain), but none have been reported for Europe, where all populations are anadromous (Renaud, 2011). The species is of conservation concern in many European countries, where management measures are taking place, especially habitat restoration and fishery management (Maitland *et al.*, 2015). It is considered a gastronomic delicacy in some European countries (e.g. Portugal, Spain, and France), and fishery are socially and economically very important there (Mateus *et al.*, 2012; Araújo *et al.*, 2016). In contrast, the sea lamprey invaded the upper Laurentian Great Lakes in the late 1910s, where it contributed, to some extent, to the collapse of lake fishery. Fishes that did not die directly from lamprey attacks, or indirectly from secondary fungal infection, had reduced market value

because of unsightly wounds. Landlocked lamprey have therefore been targeted by control measures that include lampricide treatments (Smith and Tibbles, 1980).

2.3.2 Life cycle and migrations

The anadromous sea lamprey life cycle can be divided in two distinct phases: (i) a freshwater larval phase, and (ii) a post-metamorphic marine phase. For periods of several years, the ammocoete lies burrowed in fine sediment deposits of rivers and streams, filtering from the water the micro-organisms and organic particles on which it feeds (Hardisty and Potter, 1971a; Dawson *et al.*, 2015). Almeida *et al.* (2002) found algae, primarily diatoms, were the organism most frequently found in the intestinal tract of larval anadromous sea lampreys, whereas Sutton and Bowen (1994) found that organic detritus made up 97% of the diet of the landlocked larval sea lamprey, with the remainder being composed of algae (2.2%) and bacteria (0.1%).

After a period of 3–7 years in freshwater (Beamish and Potter, 1975; Quintella *et al.*, 2003; Dawson *et al.*, 2015; Silva *et al.*, 2016a), depending on location and environmental conditions, the larva undergoes a metamorphosis. This phase is characterized by the development of the oral disk, the appearance of teeth, eruption of the eyes, enlargement of the fins, and changes in pigmentation (Hardisty and Potter, 1971b; Manzon *et al.*, 2015). For sea lamprey, metamorphosis is a requirement to prepare organisms for a life in a new habitat, the marine environment (Youson, 1980). During this phase, lampreys bear a general resemblance to the adult form, with a relatively large eye (macrophthalmia), which is characteristic of the parasitic species (Hardisty and Potter, 1971b). This stage ends with downstream migration and onset of feeding. Most internal and external changes are initiated simultaneously in midsummer. The onset is thought to be associated with sufficient lipid reserves and a change in water temperature, but not with the photoperiod (Youson *et al.*, 1993).

The period between the final transformations associated with metamorphosis (October–November) and the downstream migration to the sea, can take an average of 3–4 months in European rivers (Silva *et al.*, 2013b). Nevertheless, downstream migrants can be detected during almost the whole year. As examples: (i) in northwest Spain, the downstream migration takes place mainly between October and May, with a peak usually in March (Silva *et al.*, 2013b), and isolated migrants recorded in September and June; (ii) in Portuguese rivers, the metamorphosis and downstream migration extends from late summer (August/September) to mid-winter (January/February), with a peak in the months of October–November (Quintella *et al.*, 2003); and (iii) the downstream migration pattern of North American populations of sea lamprey has a bimodal distribution, with one peak in autumn and another in spring (Applegate and Brynildson, 1952). This latter example is the result of the climatic conditions in North America, which encourage a separation into autumn and spring migrations (i.e. onset of the winter freeze-up, and the breakup of the ice in the following spring, caused by rising temperatures and inevitably leading to high water levels; Hardisty, 2006).

The extent of the marine phase is still poorly known. Beamish (1980) proposed a period of 23–28 months between completion of metamorphosis and reproduction for this species, whereas, more recently, Silva *et al.* (2013a) suggested a shorter period of 18–20 months. Marine organisms reported to have been preyed upon by sea lamprey include bony fish, elasmobranchs, and cetaceans (Beamish, 1980; Halliday, 1991; Nichols and Hamilton, 2004; Almeida and Quintella, 2013; Silva *et al.*, 2014a). After this parasitic feeding phase in the marine environment, the sea lamprey initiates a spawning migration to continental waters where it spawns in the upstream stretches of rivers (Hardisty and Potter, 1971b; Moser *et al.*, 2015). The duration of the spawning migration varies

with the length and characteristics of the particular river system, and the location of suitable spawning sites (Hardisty, 1986b). Adult lamprey at sea are not considered to have a homing instinct (Waldman *et al.*, 2008), instead they are thought to be drawn into particular catchments by pheromones released by the ammocoetes or larvae already residing there (Bjerselius *et al.*, 2000). In addition, populations may be segregated in some regions ascribable to seabed topography isolation during the oceanic phase of the life cycle (see below; Lança *et al.*, 2014).

The timing and extent of the sea lamprey's spawning migration varies throughout its geographical range. For example: (i) in the east coast of North America, it ranges from September to March (Beamish, 1980); (ii) in the Iberian Peninsula, the spawning migration begins in December and peaks between February and March (Almeida and Quintella, 2013; Araújo *et al.*, 2016), with spawning occurring during April–June (Almeida *et al.*, 2000; Silva, 2014); (iii) in the River Severn (UK), sea lamprey migration begins in February and continues through May and June, whereas spawning occurs between the end of May and early July (Hardisty, 1986b); and (iv) in Irish rivers water temperature and volume discharge in rivers are important determining factors for sea lamprey migration and spawning success (Rooney *et al.*, 2015).

2.3.3 Stock structure and population dynamics

Anadromous sea lamprey fishing is currently present in only a few European rivers, namely in France, Spain, and Portugal. It was formerly fished extensively in the River Severn and several other UK rivers; and a historical fishery for the anadromous form existed in the 1800s on the Merrimack and Connecticut rivers, Massachusetts, before dams and pollution impacted its abundance (Renaud, 2011). Despite the former abundance of sea lamprey in the River Severn, it has been virtually extirpated from this river.

France

In Garonne and Dordogne the sea lamprey is the last migratory species exploited since 2008. Beaulaton *et al.* (2008) reported that the largest commercial fishery for sea lamprey in France occurs in the Garonne River basin, with a mean catch of 72 t over the period 1985–2003, representing > 50% of the total production in France. Over the period 1988–2003, ca. 85 000 lampreys were caught annually throughout Gironde–Garonne–Dordogne by the fishery. These captures were quite variable from year-to-year (min: 39 900; max: 154 800). In a more recent period (2000–2013) about 80 000 sea lampreys were caught annually, according to the catches reported by professional fishers of Gironde. Catches from 2008 to 2011 were above this average, whereas in more recent years they were below (about 78 000 lampreys in 2013 and 66 000 in 2014) (Plagepomi, 2015–2019). The number of lampreys caught by professional fishers, and the amount counted in control stations at dams do not always coincide. In 2000, a year when catches by professional fishers were high (154 828 kg), the number of lampreys passing the Golfech control station (Garonne) was very low (789). In 2014, an exceptionally bad year due of the late arrival of the species and the abundance observed after closure of the professional fishery, the number of lampreys was zero in both the control stations Golfech (Garonne) and Tuilières (Dordogne) (Plagepomi, 2015–2019). These numbers must be interpreted carefully, as annual fluctuations are common in diadromous fish, and may be related to several factors such as annual precipitation and river discharge. In addition, adults may concentrate downstream of these obstacles to spawn, as the Golfech and Tuilières stations are located ca. 270 and 200 km from the river mouth, respectively (Plagepomi, 2015–2019). However, a small number of ammocoetes was sampled on the Dordogne in 2014 (31 sampled locations, located downstream Tuilières

in spawning areas between Bergerac and Castillon, using electric fishing). In that year, the density of ammocoetes was much lower than in previous years, with only two 0+ larvae caught (Gracia and Caut, 2015). The small proportion of 0+ ammocoetes caught in 2014, the fact that the number of spawners detected in control stations in recent years were the lowest observed for 10 years, and the considerable catches reported by the professional fishery (100 t per year; Gracia and Caut, 2015), indicates that this system must be further monitored to assess sea lamprey population trends.

Iberian Peninsula (Spain and Portugal)

Sea lamprey is regarded as a delicacy and fetches high prices in Portugal and northwest Spain. In Spain, sea lamprey is targeted by commercial fishery in Galicia (northwest Spain), in the River Ulla, and the River Minho basins (Cobo *et al.*, 2010; Antunes *et al.*, 2015; Araújo *et al.*, 2016). Recent studies have shown that Galician rivers present the highest densities and biomasses of sea lamprey recorded in Europe (Silva *et al.*, 2016b). This study also showed signs of population increase between 2007 and 2011. However, data from more years are necessary to confirm this trend. In Portugal, annual harvest levels can be roughly estimated at 120 000–160 000 lamprey in the Minho River and 10 000–15 000 lamprey in the Tagus River (Suíças, 2010; reviewed in Maitland *et al.*, 2015 and Araújo *et al.*, 2016). Unofficial data gathered by surveying the commercial fishery in the River Mondego (central Portugal) during the 2014 spawning season estimated that around 27 000 lampreys were captured in this basin between January and April (King *et al.*, 2015). In Portugal, sea lampreys are also captured in the rivers Lima, Cávado, Douro, Vouga, and Guadiana (southern international section between Spain and Portugal) (Araújo *et al.*, 2016).

As already mentioned, sea lampreys are not considered to have a homing instinct, but instead are apparently drawn into particular catchments by pheromones released by the ammocoetes or larvae residing there. A study on sea lamprey populations in the major Portuguese river basins hypothesized the existence of three groups of sea lamprey in Portugal (North/Central group, Tagus group, and Guadiana group) based on both morphological characters and heart tissue fatty acid signature, and possibly promoted by seabed topography isolation during the oceanic phase of the life cycle (Lança *et al.*, 2014). Therefore, population assessments for *P. marinus*, along with management and conservation measures, must not be restricted to a river or watershed scale (Nunn *et al.*, 2008; Lança *et al.*, 2014; Silva *et al.*, 2016b).

Although *P. marinus* is widely distributed in Europe, the most important populations of this species, as well as the main fishery, seem to be concentrated in the southwestern regions of the continent (north-central Portugal, north-northwest of Spain, and west-southwest of France; Beaulaton *et al.*, 2008; Mateus *et al.*, 2012; Silva *et al.*, 2016b). Distinct population trends have been registered across the sea lamprey distribution range in Europe in past years (e.g. Plagepomi, 2015–2019; Gracia and Caut, 2015; Silva *et al.*, 2016b). However, more data related to longer time-series are needed to determine the actual population trend across its distributional range (Silva *et al.*, 2016b).

2.3.4 Genetics

Most molecular studies undertaken on European and North American populations of sea lamprey are based on mitochondrial markers. All of them demonstrate a lack of fixed differences in mitochondrial DNA sequences among populations of the same coast, suggesting a lack of homing (e.g. Rodríguez-Muñoz *et al.*, 2004; Waldman *et al.*, 2008), but an absence of shared haplotypes between coasts (Rodríguez-Muñoz *et al.*,

2004; Genner *et al.*, 2012). For this reason, Mateus *et al.* (2012) recommended that European and North American sea lampreys be classified as different populations, which should be managed independently.

2.3.5 International status with country highlights

International

OSPAR Convention:	Listed
EC Habitats Directive:	Annex II (except Swedish populations; 2007)
Bern Convention:	Appendix III (2002)
Bonn Convention:	Listed
Global IUCN Red List Category:	LC (NatureServe, 2013a; Freyhof and Brooks, 2011)
European IUCN Red List Category:	LC (NatureServe, 2013a; Freyhof and Brooks, 2011)
HELCOM Red List:	VU (HELCOM 2013a)
	Previous classification: EN (HELCOM 2007)

Protection and Red List status by country

Belgium (Flanders):	– / EN (Verreycken <i>et al.</i> , 2014)
Belgium (Wallonia):	– / RE (Philippart, 2007; Kestemont, 2010)
Czech Republic:	– / RE (Lusk <i>et al.</i> , 2004)
Denmark:	– / VU (HELCOM, 2013a)
Estonia:	– / – (HELCOM, 2013a)
Finland:	– / – (HELCOM, 2013a)
France:	84 Natura 2000 sites were designated for this species / NT (IUCN France <i>et al.</i> , 2010)
Germany:	– / NT (Baltic Sea) (HELCOM, 2013a)
Ireland:	Ten SACs are listed in Ireland; Fisheries Acts 1959–2006; Fisheries Act (Northern Ireland) 1966; Foyle Fisheries Act (NI) 1952; Foyle and Carlingford Fisheries Act 2007 / NT (King <i>et al.</i> , 2011)
Italy:	– / CR (Rondinini <i>et al.</i> , 2013)
Latvia:	– / – (HELCOM, 2013a)
Lithuania:	– / EN (Rašomavičius <i>et al.</i> , 2007)
Norway:	– / LC (Kålås <i>et al.</i> , 2010)
Poland:	Illegal to catch, kill or disturb / EN (HELCOM, 2013a)
Portugal:	Eight Sites of Community Importance (SCI) under the Habitats Directive / VU (Cabral <i>et al.</i> , 2005)

Russian Federation:	Included in the Red Books of St Petersburg, Leningrad District, and Russian Federation as endangered species, which means it is illegal to fish for and land this species / EN (HELCOM, 2013a)
Spain:	– / VU (Doadrio, 2001). Endangered according to decree no. 139/2011 (BOE, 2011), but only for populations from the rivers Guadiana, Guadalquivir, and Ebro, and those from the southern basins.
Sweden:	Illegal to fish for and land this species all year-round. Regional programmes for restoration of river habitats. A national action program is under development / NT (ArtDatabanken, 2015)
UK:	Six SAC rivers in England and Wales; six SAC rivers in Scotland / VU (Maitland, 2000)

2.3.6 Distribution summary

The sea lamprey is an anadromous species distributed on both sides of the North Atlantic (Figure 2.9). In North America, it occurs on the east coast from Labrador (Canada) in the north (53°N) to Florida (USA) in the south (30°N). In Europe, it can be found from the Barents Sea (Kola Peninsula, 70°N) in the north, to the Iberian Peninsula (38°N) in the southwest, and to the Adriatic Sea (40°N) in the southeast (Hardisty, 1986b). It has also been documented in the Aegean Sea (Economidis *et al.*, 1999) and the Levantine Sea (eastern Mediterranean; Cevik *et al.*, 2010). Occasionally, it occurs off Iceland, Greenland, and in the North and Baltic seas (Hardisty, 1986b). It has occasionally been found at lower latitudes in northern Africa (Boutellier, 1918; Dollfus, 1955). Several landlocked populations inhabit the North American Great Lakes, but none have been reported for Europe (Kottelat and Freyhof, 2007). In the Iberian Peninsula, it occurs in most major rivers flowing into the Cantabrian Sea and the Atlantic Ocean, as well as some into the Mediterranean (Cobo *et al.*, 2010; Mateus *et al.*, 2012).

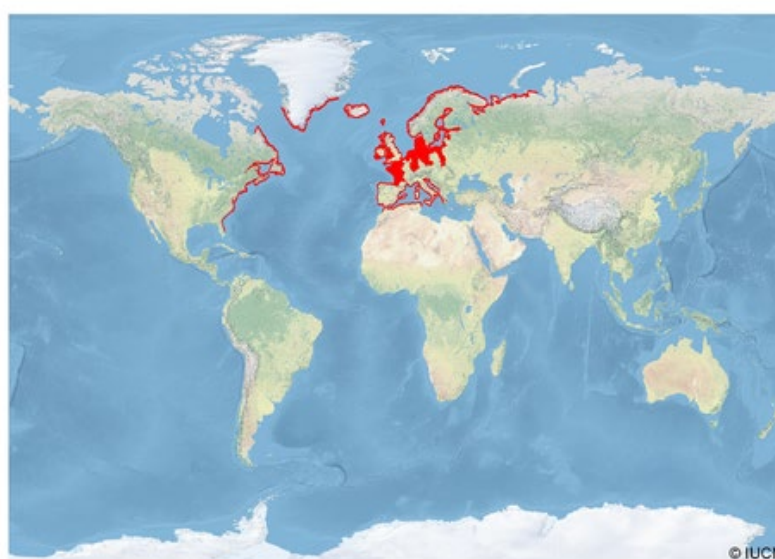


Figure 2.9. Distribution of sea lamprey (IUCN Version 2015-4).

2.3.7 Country-by-country changes in distribution and data

The sea lamprey is known to have disappeared from a number of European rivers (tables 2.7–2.9). However, there are still a number of rivers producing high catches annually, and the importance of the fishery is high in certain countries like Portugal, Spain, and France.

Ireland

The sea lamprey status was reported as Bad in its 2013 Article 17 report to the EU under the Habitats Directive (NPWS, 2013). The range of the species was small, despite substantial available habitat for spawning and nursery requirements. Major issues identified were the presence of barriers to passage in the lower reaches of designated SAC rivers (Gargan *et al.*, 2011), accumulation of spawning effort downstream of weirs, and a low-to-zero level of sea lamprey larvae in catchment-wide surveys¹¹. The impact of barriers on adult sea lamprey anadromy was also shown in a telemetry study on the River Mulkear (Rooney *et al.*, 2015). Apparent non-migratory sea lamprey have been encountered feeding on host fish in a number of Irish lakes. This was noted in the 1950s and 1960s and current ongoing investigations report the occurrence in some of the large western lakes in Ireland (Inland Fisheries Ireland, unpublished data).

Table 2.7. Historical and present distribution of sea lamprey in some countries of occurrence.

Country and river	Distribution			
	Historic		Present	
Portugal				
Minho*	X	Baldaque da Silva (1891)	X	Mateus <i>et al.</i> (2012); Lança <i>et al.</i> (2014)
Lima	X	Baldaque da Silva (1891)	X	Mateus <i>et al.</i> (2012); Lança <i>et al.</i> (2014)
Neiva	X	Baldaque da Silva (1891)	X	
Cávado	X	Baldaque da Silva (1891)	X	Mateus <i>et al.</i> (2012); Lança <i>et al.</i> (2014)
Ave	X	Baldaque da Silva (1891)	X	
Douro*	X	Baldaque da Silva (1891)	X	Mateus <i>et al.</i> (2012); Lança <i>et al.</i> (2014)
Vouga	X	Baldaque da Silva (1891)	X	Quintella (2006); Andrade <i>et al.</i> (2007); Mateus <i>et al.</i> (2012); Lança <i>et al.</i> (2014)
Mondego	X	Baldaque da Silva (1891)	X	Quintella <i>et al.</i> (2003, 2005, 2009; Mateus <i>et al.</i> (2012); Lança <i>et al.</i> (2014)
Tagus*	X	Baldaque da Silva (1891)	X	Mateus <i>et al.</i> (2012); Lança <i>et al.</i> (2014)
Guadiana*	X	Baldaque da Silva (1891)	X	Mateus <i>et al.</i> (2012); Lança <i>et al.</i> (2014)
Spain				
Bidasoa			X	Doadrio <i>et al.</i> (2011)
Oria	X	Madoz (1846)	?	

¹¹ <https://www.fisheryireland.ie/fishery-research-1/445-habitatsfull-summary-report-2013/file>

Table 2.7 (continued)

Spain (cont)	Historic distribution		Present distribution	
Asón			X	Doadrio <i>et al.</i> (2011)
Deva	X	Madoz (1846)	X	Doadrio <i>et al.</i> (2011)
Sella	X	Madoz (1846)	X	Doadrio <i>et al.</i> (2011)
Nalón	X	Madoz (1846)	X	Doadrio <i>et al.</i> (2011)
Navia	X	Madoz (1846)	X	Doadrio <i>et al.</i> (2011)
Eo	X	Madoz (1846)	X	Cobo <i>et al.</i> (2010); Silva <i>et al.</i> (2016b)
Masma			X	Cobo <i>et al.</i> (2010); Silva <i>et al.</i> (2016b)
Mera			X	Cobo <i>et al.</i> (2010); Silva <i>et al.</i> (2016b)
Ouro			X	Doadrio <i>et al.</i> (2011)
Mandeo			X	Cobo <i>et al.</i> (2010); Silva <i>et al.</i> (2016b)
Anllóns	X	Madoz (1846)	X	Doadrio <i>et al.</i> (2011); Silva <i>et al.</i> (2016b)
Tambre			X	Cobo <i>et al.</i> (2010); Silva <i>et al.</i> (2016b)
Donas			X	Cobo <i>et al.</i> (2010); Silva <i>et al.</i> (2016b)
Ulla	X	Madoz (1846)	X	Cobo <i>et al.</i> (2010); Silva <i>et al.</i> (2016b)
Umia	X	Madoz (1846)	X	Cobo <i>et al.</i> (2010); Silva <i>et al.</i> (2016b)
Lérez	X	Madoz (1846)	X	Cobo <i>et al.</i> (2010); Silva <i>et al.</i> (2016b)
Miño**	X	Madoz (1846)	X	Doadrio <i>et al.</i> (2011); Araújo <i>et al.</i> (2016)
Duero**	X	Madoz (1846)		
Guadiana**	X	Madoz (1846)	X	Doadrio <i>et al.</i> (2011)
Gualdaquivir	X	Madoz (1846)	X	Doadrio <i>et al.</i> (2011)
Guadairo		Madoz (1846)	X	Doadrio <i>et al.</i> (2011)
Guadalete			X	Doadrio <i>et al.</i> (2011)
Ebro	X	Madoz (1846)	X	Doadrio <i>et al.</i> (2011)
Llobregat	X	Madoz (1846)	?	
France				
System Gironde-Garonne-Dordogne	X	Gracia and Caut (2015)	X	Gracia and Caut (2015)
Livenne	?		X	Taverny <i>et al.</i> (2011); Gracia and Caut (2015)
Dronne	?		X	Taverny <i>et al.</i> (2011); Gracia and Caut (2015)

Table 2.7 (continued)

France (cont)	Historic distribution		Present distribution	
Cère	?		X	Taverny <i>et al.</i> (2011); Cazeneuve and Lascaux (2008)
Dordogne	?		X	Taverny <i>et al.</i> (2011)
Adour	?		X	Beaulaton <i>et al.</i> (2008)
Loire	?		X	Beaulaton <i>et al.</i> (2008)
Scorff	?		X	Sabatié (1998)
Ciron	?		X	Gracia and Caut (2015)
Vézère	?		X	Plagepomi (2015–2019)
Levre	?		X	Plagepomi (2015–2019)
Ireland				
Boyne	X	Kurz and Costello (1999)	X	NPWS (2013)
Liffey	X	1906, 1977 National Museum	X	
Avoca			X	
Slaney	X	Kurz and Costello (1999)	X	NPWS (2013)
Barrow	X	Kurz and Costello (1999)	X	NPWS (2013)
Nore	X	Kurz and Costello (1999)	X	Gargan <i>et al.</i> (2011); NPWS (2013)
Suir	X	Kurz and Costello (1999)	X	NPWS (2013)
Munster Blackwater	X	Kurz and Costello (1999)	X	NPWS (2013)
Lee	X	Kurz and Costello (1999)	X	
Laune	X	1969 Kurz and Costello (1999)	X	NPWS (2013)
Feale			X	NPWS (2013)
Mulkear	X	Kurz and Costello (1999)	X	NPWS (2013); Rooney <i>et al.</i> (2015)
Shannon	X	Kurz and Costello (1999)	X	NPWS (2013)
Fergus	X	Kurz and Costello (1999)	X	NPWS (2013)
Corrib	X	1893, 1943 National Museum; Kurz and Costello (1999)	X	NPWS (2013)
Moy	X	1929 Kurz and Costello (1999)	X	NPWS (2013)
Garavogue	X	Kurz and Costello (1999)		
Poland				
Vistula basin	?	Witkowski (2010)	?	Witkowski (2010); inquires conducted by NMFRI employees
Oder basin	?	Witkowski (2010)	X	Mariusz Raczyński (pers. comm.); inquires conducted by NMFRI employees
Polish coastal rivers	?	Witkowski (2010)	?	Witkowski (2010); inquires conducted by NMFRI employees

Table 2.7 (continued)

Sweden	Historic distribution		Present distribution	
Göta älv	X	1890	X	Ljunggren and Söderman (2016)
Örekilsälven	X	1987	X	"
Ennindalsälven	?		X	"
Kungsbackaån	X	1906	X	"
Viskan	X	1988	X	"
Rolfsån	X	1960s	X	"
Löftaån	X	1960s	X	"
Ätran	X	1940s	X	"
Suseån	?		X	"
Nissan	X	1980s	X	"
Fylleån	X	1940s	X	"
Genevadsån	X	1960s	X	"
Lagan	X	1970s	X	"
Stensån	?		X	"
Rönneå	?		X	"
Skräbeån	?		X	"
Råån	?		X	"
Mörrumsån	?		X	"

* Presence confirmed on the Portuguese side; ** Presence confirmed on the Spanish side.

Portugal

Sea lampreys were historically reported in Minho, Lima, Neiva, Cávado, Ave, Douro, Vouga, Mondego, Tagus, and Guadiana basins (Baldaque da Silva, 1891), but have since disappeared from some of these basins (e.g. the River Ave). In the information provided in the national report under Article 17 of the Habitats Directive by the Portuguese Environmental Agency, covering the period 2007–2012, *P. marinus* was classified as "U1=" (unfavourable inadequate - stable) in the Mediterranean biogeographical region (MED), the same classification attributed by the member state for the period 2001–2006. As for the Atlantic biogeographical region (ATL), *P. marinus* was classified as "FV" (Favourable) in the period 2007–2012, whereas in the previous report (period 2001–2006), it was classified as U1 - Unfavourable inadequate. The change observed was attributed to the existence of more accurate data (e.g. better mapping of distribution) or improved knowledge (e.g. on ecology of species or habitat). Population evaluation was "XX" (unknown) for both regions; range evaluation was FV for both regions; habitat evaluation was FV for ATL, and U1 for MED; and future perspectives was FV for ATL, and U1 for MED¹².

Spain

The species is present in almost all the main coastal rivers of the northwest and some in the north (Cobo *et al.*, 2010; Doadrio *et al.*, 2011; Silva *et al.*, 2016b). It is much less

¹² available at: <http://www2.icnf.pt/portal/pn/biodiversidade/rn2000/dir-ave-habit/rel-nac/rel-nac-07-12>

common in the south and the east, with populations in the rivers Guadiana, Guadalquivir, Guadairo, Guadalete, and Ebro (Doadrio *et al.*, 2011). Sea lamprey fishery were present in 52 rivers within 22 provinces of Spain in the first half of the 19th century, whereas now sea lamprey are only captured in two rivers and two provinces (Madoz, 1846; Araújo *et al.*, 2016). The main causes have been the dramatic reduction and degradation in habitat, pollution, and fishery. Habitat accessible for sea lamprey in north-west Spain has been reduced to 36% of the total river length, on average (Silva *et al.*, 2016b). For the whole Iberian Peninsula, the accessible habitat has been estimated to be ca. 20% of the total river length (Mateus *et al.*, 2012).

France

Populations from the Garonne–Dordogne system need special attention regarding fishery management, according to data from ammocoete abundance surveys and spawner counts in the control stations in dams in recent years (see above).

Table 2.8. International conservation status and legislation (based on the Habitats Directive, Bern Convention, CITES, IUCN and conservation status by country, with a comparison between the status reported for sea lamprey by the Study Group on the Status of Diadromous Fish Species (SGSDFS) in 2005 (ICES, 2005) and the current situation.

Criteria		
International classification		
Habitats Directive	Annex II	
CITES	-	
Bern Convention	Annex III	
IUCN V 2015-4	LC (Population trend: stable)	
National classification		
Country	2005	Last assessment
Denmark	No category assigned	VU (HELCOM, 2013a)
Finland	No category assigned	No category assigned
France	VU	NT (IUCN France <i>et al.</i> , 2010)
Belgium - Flanders	EX	EN (Verreycken <i>et al.</i> , 2014)
Germany	EN	Not threatened (Baltic Sea) (HELCOM, 2013a)
Ireland	Indeterminate	NT (King <i>et al.</i> , 2011)
Italy	CR	CR (Rondinini <i>et al.</i> , 2013)
Norway	No category assigned	LC (Kålås <i>et al.</i> , 2010)
Poland	No category assigned	EN (HELCOM, 2013a)
Portugal	No category assigned	VU (Cabral <i>et al.</i> , 2005)
Spain	VU	VU (Doadrio, 2001)*
Sweden	VU	NT (ArtDatabanken, 2015)
UK	No category assigned	VU (Maitland, 2000)

DD: Data Deficient; VU: Vulnerable; NT: Near Threatened; EN: endangered; CR: Critically endangered; RE: regionally extinct; EX: extinct; NA: not applicable

*Endangered according to decree no. 139/2011 (BOE, 2011), but only for populations from the rivers Guadiana, Guadalquivir, and Ebro, and those from the southern basins.

Table 2.9. Sea Lamprey monitoring programmes and data in some countries of occurrence

Country	Monitoring programme	Available data
Portugal	1998–present: monitoring of sea lamprey larvae and adults in River Vouga	Surveys for larvae abundance using electric fishing; radio telemetry; physiological telemetry (EMG) (e.g. Quintella, 2006; Andrade <i>et al.</i> , 2007)
	1998–present: monitoring of sea lamprey larvae and adults in River Mondego	Surveys for larvae abundance using electric fishing; radio telemetry; EMG; PIT tags (e.g. Quintella <i>et al.</i> , 2003, 2005, 2009)
	2011–present: monitoring of the fish pass at the Açude-Ponte dam, River Mondego, Coimbra	PIT tags, radio and acoustic telemetry, EMG; spawners counting system; surveys for larvae abundance; surveys conducted by professional fishers (Almeida <i>et al.</i> , 2015; Pereira <i>et al.</i> , 2016)
	2012–present: surveys conducted by professional fishers in main river basins	Number of lampreys captured, fishing gears used and fishing effort per gear
Spain	2007–2011: monitoring of larval sea lamprey populations in Galician rivers	Monitored rivers: Eo, Masma, Mera, Ouro, Anllóns, Mandeo, Tambre, Donas, Ulla, Sar (Ulla tributary), Umia, and Lerez (Silva <i>et al.</i> , 2016a); method: single pass of electric fishing (Silva <i>et al.</i> , 2014b).
	Captures declared in fishery	River Ulla from 2003 to present (Araújo <i>et al.</i> , 2016); River Minho basin from 1914 to present (Araújo <i>et al.</i> , 2016)
	Fish pass survey in the River Ulla	Captures of sea lamprey registered from 1997, both adults during their upstream migration and post-metamorphics during downstream migration (Silva <i>et al.</i> , 2013a, 2013b, 2016a)
Ireland	2013–2018: catchment-wide larval surveys in SAC catchments; 2000–2015: occasional sea lamprey redd count surveys in SACs	Reports available at: https://www.fisheryireland.ie/Projects/habitats-directive-and-red-data-book-fish-species.html
Poland	---	Extremely rarely noted in fish traps (commercial fishery)
Sweden	Fish pass surveys	>10 Vaki counters with cameras from ca. 2013 onward

Other countries

In Denmark, sea lamprey it is classified as VU, and there are indications of decline since 1990. Swedish inventories in Kattegat show reproduction in eight rivers, with an estimated total of 800 reproducing individuals in 2008. Sea lamprey is caught irregularly, but almost annually in Estonia, whereas in Finland, Russian Federation, and Latvia, the species is not an annual catch. For example, it has been reported only eight times since 1927 in the Russian Federation part of the Gulf of Finland. In Poland, there may be a spawning population in the Oder River, but this has not been verified (HELCOM, 2013a). Sea lamprey is caught regularly in the Arkona Basin, but no sea lamprey reproduction site is known from rivers of the German Baltic Sea area (Thiel *et al.*, 2009).

2.3.8 Threats

The main threats identified for sea lamprey are the construction of dams and weirs in rivers and degraded water quality. In the River Ave in northern Portugal, populations have now vanished, probably owing to pollution (Quintella, 2006; Mateus *et al.*, 2012). Industrial pollution is probably also responsible for the extremely low density of sea lamprey larvae populations in the lower reaches of the River Cávado (Almeida *et al.*, 2008). Dams and weirs block upstream migration to spawning grounds and reduce habitat available for the larval stage. During the months of upstream migration and reproduction, lamprey do not feed, and a dramatic decline in energy reserves occurs in the adults (Barca *et al.*, 2015; Silva *et al.*, 2016a). The presence of obstacles increases the expenditure of energy during the upstream migration, leaving less resources available for reproduction (Lucas *et al.*, 2009; Quintella *et al.*, 2009). Overfishing may also be affecting this species in some areas where it is still commercially important (e.g. Portugal, Spain, and France). These areas also hold the main populations of the species (Mateus *et al.*, 2012; Silva *et al.*, 2016b). However, there is insufficient reliable data on the exploitation rate of sea lamprey populations at those fishery to be able to identify and quantify possible overexploitation. This prevents, in most basins, a proper fishery management to ensure sustainable exploitation (Araújo *et al.*, 2016).

2.3.9 Conclusion - future

There are two opposite scenarios regarding management of sea lamprey across its distributional range. In the American Great Lakes, the sea lamprey is invasive, and the Canadian and US governments spend large amounts of resources each year trying to control this species. In contrast, many countries in Europe are trying to conserve populations or, at least, manage sea lamprey for fishery. In their native North American ranges, sea lamprey are largely understudied and undermanaged.

In Europe, the main cause for population decline is habitat degradation, such as river damming and pollution, which is exacerbated by the effect of fishing pressure in some regions. Considering the conservation status of sea lamprey in the countries holding the main populations (i.e. France, Spain, and Portugal), it is felt that the classification of the European populations of *Petromyzon marinus* should be revised from its current classification as LC in the European Red List of freshwater fish. It is proposed that sea lamprey be rather classified as VU, because it is included within this category in important areas of its distribution.

Management recommendations:

- Habitat restoration: (i) barriers to migration should be equipped with suitable fish passes, and (ii) water quality should be monitored and improved.
- Maintain sustainable fishery and stocks: legislation should be carefully revised in all countries holding sea lamprey fishery, and discussions with professional fishers should be increased.

2.4 Sea trout (anadromous brown trout)

2.4.1 Taxonomy

Class:	Actinopterygii
Order:	Salmoniformes
Family:	Salmonidae
Scientific name:	<i>Salmo trutta</i>

Subspecies, variations, synonyms

None

Taxonomic notes

None

Common names

EN: Trout; DE: Meerforelle; DA: Havørred; ES: Meriforell; FR: Truite de mer; FI: Taimen; LV: Taimiņš; LI: Šlakis; PL: Troć; RU: Kumzha, forel; SV: Havsöring; PT: Truta-marisca

General characteristics

Adult length:	20–140 cm
Weight:	0.075–50 kg
Maximum age:	38 years (www.fishbase.org)
Generation length:	2–8 years on average

Definitions

Parr:	A juvenile salmonid. Commonly defined as the period after the yolk sac has been fully absorbed, but before smoltification (Elliott, 1994)
Sea trout:	The anadromous form of the brown trout.
Smolt:	A parr that has undergone morphological, behavioural, and physiological changes that enables it to migrate into saline environments, but still reside in freshwater.
Post-smolt:	A smolt that has entered seawater.
Finnock:	A small sea trout in its first year after smolt migration. Other, regional, names include whitling and school peal.
Kelt:	A sea trout after spawning and before it returns to the sea.
Veteran migrant:	A sea trout that has completed a migration cycle from the river and back. This includes kelts and non-spawning individuals.

The native distribution of brown trout is restricted to Europe (Figure 2.10). However, through stocking into rivers in many parts of the world, the species has been spread to ca. 50 countries, starting with eastern Russia in 1852. The brown trout is well known for its wide range of life-history strategies. Different populations range from spending their entire lifespan in the freshwater environments of lakes, rivers, and streams (resident and lake-migratory populations) to performing long-distance migrations between freshwater and marine habitats (sea-migratory or anadromous populations).

This flexibility is probably a contributing explanation to its successful colonization of new areas. Sea trout is the fully anadromous form of the species, and they usually co-exists in freshwater with resident brown trout, as part of the same breeding population (i.e. one part of the population, predominantly female, leaves the river for feeding in the sea, whereas another part stays in the river as residents (Elliot, 1994)). The availability in streams of suitable spawning areas for the adult fish, and nursery habitat for the juvenile fish is an important determinant of population strength, which, in turn, generally sets the population size limit (Chapman, 1966; Elliot, 1994).

Sea trout is a popular target for sportfishing in both sea and rivers. In addition, sea trout are caught in nets and traps operated by non-professionals. The fish can be the primary target for the fishery, or it can be caught as bycatch in a fishery for other species. The recreational catch is known, although with limited and varying accuracy. The share of the catch taken by recreational fishers has increased in recent years, in some areas being several orders of magnitude larger than the commercial catch.



Figure 2.10. Distribution of sea trout.

2.4.2 Life cycle and migrations

Sea trout in the northern hemisphere usually spawn during autumn in rivers and smaller streams, often in the upper reaches or in smaller tributaries, where females excavate a gravel nest prior to mating (Elliott, 1994). The largest and most dominant males usually defend the females and nesting sites with the highest quality (Klemetsen *et al.*, 2003). Subordinate males may instead adopt an alternative sexual strategy, called “sneaky-mating”, whereby they attempt to fertilize some of the eggs before the dominant male (Webb *et al.*, 2007). Eggs are generally distributed in two or three nests, and their number ranges from 100 for a small resident trout (Elliott, 1994), to several thousand (Webb *et al.*, 2007). Both sexes usually survive spawning, and anadromous trout

either migrate back to the sea or a lake in autumn, or overwinter in rivers and migrate out in spring.

The eggs hatch the subsequent spring (February–April) (Elliott, 1994), but the juveniles (alevins) remain in the sheltered gravel nest for ca. 5–8 weeks feeding endogenously on their internal energy store (yolk sac) (Webb *et al.*, 2007). As the yolk-sac supply diminishes, the “fry” emerge to the gravel surface usually between March (Spain) and July (Finland). As fry emerge, they start feeding on invertebrate prey (Skoglund and Barlaup, 2006). During this critical period, they compete intensively for feeding territories near the spawning area. A large proportion will drift downstream, where they will experience increased mortality rates as a result of starvation and predation (Elliott, 1989). The limited supply of feeding territories at emergence (Nislow *et al.*, 1998) imposes a strong selective pressure on early emergence (prior residency; Harwood *et al.*, 2003) and body size at emergence (Good *et al.*, 2001). These two features are, to a large extent, maternally determined (Einum and Fleming, 2000). During the “parr” phase, the juveniles generally develop the characteristic red spots and vertical stripes on the sides of the body (Webb *et al.*, 2007). As individuals grow, territoriality can become replaced by a more flexible behaviour, where the parr use home ranges and form dominance hierarchies (Keeley, 2000). Although debated (Gowan *et al.*, 1994; Rodriguez, 2002), stream-living salmonids seem to be relatively sedentary after emergence, with movement distances rarely exceeding 200 m (e.g. Heggenes, 1988; Bohlin *et al.*, 2002; Steingrímsson and Grant, 2003; Økland *et al.*, 2004).

Individuals adopting an anadromous life strategy migrate to the ocean from their juvenile habitats in freshwater streams at 1–5 years of age, and return to the freshwater habitat as sexually mature adults (Klemetsen *et al.*, 2003; Milner *et al.*, 2003) after a period of ½–5 years foraging in the sea. Survival during this migration is highly dependent on the distance the fish needs to travel, and the kind of environment through which it has to pass. For example, passing through lakes and weirs can be problematic, and is associated with very serious mortality rates (Aarestrup and Koed, 2003; Greenberg *et al.*, 2012; Schwinn *et al.*, 2017). The initial phase in the sea (post-smolt stage) is also considered critical, whereas survival during the growing phase until maturity is much better.

The underlying mechanism that determines whether the sea trout will migrate or become resident is primarily based on the growth trajectory of the parr in the preceding summer. If a certain threshold for migration is reached, the individuals will maintain their appetite and continue to grow. Meanwhile, individuals failing to reach the threshold will slow their growth. This results in two life-history patterns: migratory and residency (Økland *et al.*, 1993; Bohlin *et al.*, 1996; Thorpe *et al.*, 1998; Rikardsen *et al.*, 2004). In addition, there is a reproductive threshold prior to spawning in autumn for individuals of both life-history patterns, which is based on the lipid content trajectory. If they are above this threshold, they will start investing in reproductive tissue. It has been suggested that an individual that had previously decided to migrate can change this decision and start investing in reproductive tissue instead, thereby abandoning its migratory tactic (Thorpe *et al.*, 1998). Individuals that fail to reach the maturation threshold remain immature, and the fastest growers continue on their migration pathway. This suggests that the largest individuals are the ones that migrate. However, most studies instead suggest that growth potential is the governing factor (Økland *et al.*, 1993; Bohlin *et al.*, 1996). The migrating sea trout must then adapt for a life in the sea (smoltify) (Thorstad *et al.*, 2016). This transformation includes both physiological and morphological changes (McCormick *et al.*, 1987), and is generally induced by photoperiod and temperature (Björnsson *et al.*, 2011).

While at sea, trout generally undertake shorter migrations than salmon (*Salmo salar*), although some individuals have been recorded migrating hundreds of km (Degerman *et al.*, 2012). Many fish return to freshwater as “whitling” or “finnock” after spending only one summer at sea (Allan and Ritter, 1977). The underlying mechanism for this early return is generally poorly understood. In some studies, these returning whitlings/finnock have been reported to stay in freshwater to mature and migrate to their spawning sites. There they have been reported to reach a proportion of up to 30% of spawners (Davidson *et al.*, 2006), and conduct a similar sneaky mating tactic to the one carried out by resident males. Earlier studies, in contrast, stated that the majority of these fish are immature, and that the migration is not linked to reproduction (Le Cren, 1985). Instead, it has been hypothesized that a combination of high salinity and low water temperatures may force these individuals to overwinter in freshwater (Thomsen *et al.*, 2007).

During upriver migrations, the silvery colour of sea trout evolves into a dark breeding colour, the skull of males enlarges, and the lower jaw in males develops an upturned kype (Fleming, 1996).

2.4.3 Stock structure and population dynamics

Generally, there is little information on stock structure and population dynamics in anadromous sea trout. Tagging studies indicate that most mature individuals return to their natal streams (Elliott, 1994). Sea trout in the Baltic have been reported to have a pronounced substructuring, with markedly divergent populations distinct between and within rivers, streams, and creeks: small effective sizes in individual creeks, meta-population structure over moderately sized areas (e.g. the islands of Gotland and Bornholm), and a small but clear component of isolation by distance observed in some areas of the Baltic (e.g. Laikre *et al.*, 2005; Koljonen *et al.*, 2014).

2.4.4 Genetics

Sea trout harbour extensive genetic diversity (Keller *et al.*, 2011), which is expected to promote local adaptation. Physical isolation (populations living in inaccessible parts of a stream or a cut-off lake system) and homing of the species should contribute to a high genetic variation among populations. However, adaptive divergence among populations may be hampered by potentially high levels of gene flow owing to natural straying and/or human management practices. Surprisingly little quantitative information exists about homing behaviour in sea trout, but it is generally believed that compared with the salmon, sea trout are less adapted to a specific river and a large degree of straying occurs (Berg and Berg, 1987; Degerman *et al.*, 2012). There are, however, several reports throughout Europe suggesting that sea trout form local, genetically differentiated adapted populations (Taylor, 1991; Hansen *et al.*, 2007; Fraser *et al.*, 2011; Meier *et al.*, 2014), whereas recently, there are also a number of studies suggesting a close genetic relationship among neighbouring populations forming distinct groups. Quéméré *et al.* (2015) detected hierarchical genetic structure with two main genetic units corresponding to the eastern and western marine ecoregions of the English Channel delineated by the Cotentin peninsula, and Höjesjö *et al.* (in prep.) detected a genetic population structure along the coastline of Sweden. Hence, this suggests that at least a relatively large degree of straying probably occurs, resulting in local adaptations within a broader geographical range rather than to a single river system, even if this needs to be further evaluated.

2.4.5 International status with country highlights

International

OSPAR Convention:	Not listed
EC Habitats Directive:	Not listed
CITES:	Not listed
Bern Convention:	Not listed
Bonn Convention:	Not listed
IUCN Criteria:	–
Global IUCN Red List Category	LC
European IUCN Red List Category:	LC
HELCOM Red List:	VU

National

Denmark:	Fishery regulations. Fishing with net prohibited up to 100 m from shoreline, closed areas around river mouths (time depending on the size of river). Minimum landing size 40 cm. Mature fish protected 15 November – 15 January. Regular stockings / LC.
Estonia:	Regular stockings. Fishery regulations / NT.
Finland:	In most waters, wild-born trout are not allowed to be caught at all (intact adipose fin; stocked fish are marked by cutting of adipose fin). Fishing is not allowed during spawning time in freshwater. Minimum legal landing size for wild-born trout is 60 cm (in waters where fishing is still allowed). Fishing restriction areas outside some river mouths. Regular stockings / CR (sea migrating).
France:	Sea trout status is not formally assessed on a national basis. The minimal size for capture is 35 cm (fork length) both at sea and in rivers. Some small professional fishery targeting salmonids exist, but the biggest part of the captures is by anglers and recreational net fishers.
Germany:	Minimum landing size 40 or 45 cm TL, and different closed seasons exist from mid-autumn to early winter depending on Federal state. Net fishing prohibited within 200 m from the shoreline (one Federal state) and protected zones exist in river mouths during spawning migration in late autumn and early winter. A restocking programme has been established since the early 1980s, with ca. 1 million fry being released in suitable rivers every year since 2000 / NT (Baltic Sea).
Ireland:	Sea trout > 40 cm fork length are classified as salmon under national legislation and are covered under salmon control regulations. There is a very limited commercial fishery for sea trout, and the primary harvesting is from angling. Commercial and rod harvest of salmon is permitted where stocks are in surplus (exceeding a system-specific salmon conservation limit), and salmon and sea

trout fishery are very strictly controlled. River-specific harvest management advice is provided for salmon annually. Approximately one-third (circa 60 systems per annum) of Irish salmon and sea trout fishery are open for harvest, with the remainder being open for catch-and-release angling, or closed to angling. (EU legislation status: Unprotected, IUCN Red List: LC).

Latvia:	Trout are legislated under the law on the conservation of species and biotopes; included in CM regulation Nr. 396 and 45. Protection by commercial fishing and angling rules (closed season, minimal landing size) / –.
Lithuania:	Restoration programme in some rivers. Protected from fishing during spawning time in rivers from 1 October to 31 December. Restricted fishery in migration routes / LC.
Norway:	Sea trout at sea are protected in some regions from 1 March to 1 April. There are general fishing restrictions in areas outside some river mouths. For remaining river mouths, fishing outside the fishing season is prohibited. In watercourses, there are local regulations for fishing season and minimum landing size. In Finnmark, Troms, and Nordland, the minimum landing size is 30 cm, whereas southward the rest of the country has a minimum landing size limit of 35 cm.
Poland:	Stocked annually in all rivers. Minimum landing size 50 cm. Closed season different in coastal and open waters. Minimum mesh size 70 mm (bar length) for fishery in the coastal area of Gdańsk Bay, 80 mm for the rest of Polish maritime areas. Protected area (closed for fishery) in the river mouths. In Polish “Red book” sea trout is stated as LC / cd (conservation dependent).
Portugal:	The anadromous form of the species is considered CR in the last revision of the Portuguese Red List of Threatened Vertebrates (Cabrál <i>et al.</i> , 2005). Sea trout is also included in Portuguese legislation related with fishing activities, within the jurisdiction of both inland and marine fishery authorities.
Russian Federation:	The anadromous form is included in the Red Books of St Petersburg, Leningrad District, and Russian Federation, which means it is illegal to fish for and land this species / EN.
Spain:	In the Spanish legislation, <i>Salmo trutta</i> is not included on any protection list or endangered category, but this species is cited as “vulnerable” in the Red Book of Spanish Vertebrates (1992). In regional regulation of the different Spanish regional governments (CCAA), <i>Salmo trutta</i> is included in a regulation of the Navarre Government (1995), in which the species is classified as of “special interest”.
Sweden:	Protected from fishing during spawning time in some areas. Minimum legal landing size is 40 cm in Åland Sea, 45 cm in Kattegat, and 50 cm in the rest of the Baltic Sea / LC.
UK (England and Wales):	No national listing, but typically treated as a salmon in national and local legislation.

UK (Northern Ireland): None listed for trout in the region.

UK (Scotland): Currently no formal assessments of the status.

2.4.6 Distribution summary

Range: Sea trout is the anadromous form of brown trout (*Salmo trutta* L.), naturally distributed in the Atlantic, Channel, North, White and Baltic seas basins from Spain to Chosha Bay (Russia) as well as in Iceland (Section 2.4.7). In the Rhône drainage, sea trout is native only to the Lake Geneva basin, which it entered after the last glaciation. Sea trout is also native to the upper Danube and Volga drainages.

Native: Andorra, Austria, Belarus, Belgium, Croatia, Czech Republic, Denmark, Estonia, Faroe Islands, Finland, France, Germany, Guernsey (UK), Hungary, Iceland, Ireland, Isle of Man (UK), Jersey (UK), Latvia, Liechtenstein, Lithuania, Luxembourg, Netherlands, Norway, Poland, Portugal, Romania, Russian Federation, Serbia, Slovakia, Slovenia, Spain, Sweden, Switzerland, Ukraine, and UK.

Introduced: throughout Europe, North and South America, southern and mountainous eastern Africa, Kerguelen Island (French subAntarctic territories), Pakistan, India, Nepal, Japan, New Zealand, and Australia.

2.4.7 Country-by-country changes in distribution

Denmark

The coastline of Denmark is 7314 km long and, historically, it held an estimated 800 sea trout streams. The number of streams with sea trout has increased in recent decades from a low of 176 in 1960 to 406 in 2012. Many very small streams flowing directly to the coast have viable populations of sea trout. Trout populations are monitored through an extensive network of stream stations, electrofished on average every 7 years. The densities of 0+ trout have increased significantly from 20.8 fish 100 m⁻² in the 1980s to 59.7 fish 100 m⁻² in the 2010s, although regional differences are large. It is currently estimated that the natural smolt production in Denmark is about 600 000, having increased from around 200 000 in the 1980s.

Finland

The coastline of Finland is 1100 km long (or 4600 km if inlets are included), with most of the coast sheltered by archipelagos. The historical number of sea trout rivers is unknown, but sea trout currently occur in 28 rivers. In addition, there are 79 potential rivers and brooks. Sea trout populations suffer from overexploitation, and degradation of river habitats owing to dredging, damming, sulphate soil, and eutrophication. The restoration of nursery habitats and building of fishways has occurred in a number of rivers in the last 20 years. According to IUCN criteria, the status of sea trout is CR. Fishing pressure is still too high, mainly as a consequence of coastal net fishing targeting other fish species, in which sea trout are caught as bycatch. However, as a result of stricter fishing regulation at sea, parr densities have increased in the last few years in several rivers, particularly in the Gulf of Finland area. A national salmon and sea trout strategy was accepted in 2014, and regional plans for improving sea trout stocks are under compilation.

France

The largest sea trout populations are located in northern areas. Along the 4853-km coastline of France, ca. 100 rivers are officially listed as sea trout rivers, according to French legislation. However, the true number is definitely higher, owing to the fact that many small rivers are not taken into account. Sea trout status in those rivers varies from locally vulnerable to positive, as a consequence of a national strategy to restore ecological continuity. No major change in sea trout distribution has been documented.

Germany

Sea trout are widely distributed in rivers flowing into the Baltic Sea. About 36 German Baltic river populations are supported by annual releases of fry or yearling fish, originating in wild-caught spawning individuals. In 2015, 1.2 million fry and about 13 500 smolts were released into Baltic rivers. About 10–25 rivers contain exclusively wild, or at least almost wild, self-sustaining populations. The number of rivers with sea trout has increased over the last 25 years. Programmes vary considerably, both in spatial and temporal resolution, because sea trout are managed at the Federal state-level monitoring. The existing time-series are too short to extract clear patterns. However, 2014 showed significantly above-average abundance in parr numbers in many surveyed Baltic Sea rivers. This coincides with an all-time high in spawning run counts measured by video-camera monitoring from autumn 2013 to early winter 2014 in one river, the Hellbach. Parr abundance status has been variable in the last three years. According to ICES WGBAST (ICES, 2015b) methodology (taking the Trout Habitat Score or THS as a measure of habitat quality for 0+ and 1+ parr into consideration), ca. 40–80% of the potential parr production is reached. Some small rivers produce more.

Ireland

Sea trout are widely distributed, occurring in all coastal and estuarine waters, and spawning populations are present in the majority of rivers (Table A3.1, Annex 3). It is estimated that 256 named river systems have sea trout populations (IFI, unpublished). The majority of these systems enter the sea and/or estuarine waters directly, and ca. 148 also support Atlantic salmon populations. An assessment (McGinnity *et al.*, 2003), primarily focused on salmon, classified 88 river systems as “sea trout only” rivers that do not have significant salmon populations (McGinnity *et al.*, 2003; NASCO, 2005). A large proportion of sea trout stocks in Ireland, particularly in the west of the country, are consists of finnock. Although direct monitoring of sea trout stocks is not systematically undertaken in all 256 sea trout systems, there is no evidence (from rod catch data, surveys, fishery staff observations, or angling reports) to suggest that historical and present distributions have changed, despite periodic fluctuations in population abundance in individual systems (Table A3.2, Annex 3). However, in two systems in the west of Ireland (Owengowla and Invermore), only remnants of a sea trout population remain, after sustained marine mortalities of smolts coinciding with the operation of marine salmon farms (Gargan *et al.*, 2006a). A substantial sea trout stock collapse was documented for many river systems in the west of Ireland in the late 1980s (reviewed in Gargan *et al.*, 2006b). This is reflected in the rod catch data, available for 18 Connemara sea trout fishery in western Ireland since 1974 (Figure A3.1, Annex 3). The data show an annual catch of about 10 000 sea trout up to the mid-1980s followed by a catch collapse over the 1989/1990 period. Post 1990, angling in these fishery has been on a catch-and-release basis, but data show that the sea trout rod catch has not recovered to historical levels (reviewed in Gargan *et al.*, 2006b; IFI, unpublished).

Norway

The coastline of Norway is 25 148 km long and extends over a broad latitudinal range. Sea trout populations occur all along this coast. There has been reported catch of sea trout in ca. 500 Norwegian watercourses during 1993–2014 (Anon., 2015). The sea trout stocks in southern and eastern Norway are relatively stable. There is a general decline in western Norway and the middle part of Norway, whereas catch statistics in northern Norway are relatively stable. A total of 1127 streams support wild populations of sea trout, of which 137 are threatened or vulnerable, and 21 have been lost. Through the work undertaken in the Norwegian Scientific Advisory Committee for Atlantic salmon (www.vitenskapsradet.no), a classification system and a classification of sea trout along the Norwegian coast is in progress.

Poland

Sea trout is abundant in 25 rivers (Figure A3.2, Annex 3). Almost all populations are supported by stocking, and the true status of wild recruiting sea trout is difficult to assess. All populations are considered mixed, based on wild recruitment and stocking. The main decreases in distribution occurred at the beginning of the 20th century, following the damming of the majority of rivers in northern Poland, and in the 1960s after southern Poland was cut off by the big dam in the middle run of the Vistula River. Improvement in water quality in Polish rivers since the beginning of the 1990s, and some new fish passes built recently, have resulted in an increase in some sea trout populations.

Portugal

Occurrence is limited to northern and central regions of Portugal (Figure A3.3, Annex 3). Potentially, sea trout might also occur in the river basins of the Douro, Vouga, and Tagus (probably only in the northern part of the catchment). The sea trout population from River Minho is probably the largest in Portugal (Cabral et al., 2005). However, studies on sea trout abundance and state have not yet been developed for other rivers where its presence has been recorded, aside from the Lima basins (Maia, 2003). Data from professional and recreational fishers operating in rivers Minho and Lima indicate that the number of sea trout adults in these two river basins is extremely low (i.e. reduction may have affected 98% of sea trout adults in the past 10–15 years). Populations of anadromous trout seem to be in an accentuated decline in most of their occurrence area in Portugal (Cabral et al., 2005).

Spain

Sea trout is only present in rivers that drain to the Atlantic Ocean and the Cantabrian Sea, in the north and northwest area. Sea trout is present from the Minho River, bordering with Portugal, to the Bidasoa River, bordering with France. Sea trout fishery management is the responsibility of regional governments (known as Autonomous Communities = CCAA). The five CCAA involved in sea trout management, from east to west, are: Navarre, Basque Country, Cantabria, Asturias, and Galicia (Annex 3, figures A3.3 and A3.4). A survey of sea trout managers of the five CCAA showed that sea trout was present in 48 of 52 rivers analysed (all the rivers longer than 15 km in the area). Absence in the remaining four rivers was caused by water pollution in three of them, and an impassable natural waterfall at the mouth of the river in the other. Of the ca. 3000 km length of the 48 main courses where trout are present, only 40% (1200 km) is accessible to anadromous fish (figures A3.4 and A3.5, Annex 3). This reduced accessibility has been caused mainly by hydroelectric dam construction in the second half of the 20th century. As well as the 48 rivers analysed, there are several small coastal

rivers in the Spanish sea trout area that accommodate very interesting, and probably not very exploited, sea trout populations. Annex 3, Figure A3.6, shows the sea trout distribution as recorded in the Atlas and Red Book of the Freshwater Fish from Spain (Doadrio, 2001).

Sweden

The coastline of Sweden is ca. 2400 km long, and sea trout occur all along this coast. At least 800 individual rivers and streams support sea trout populations (Degerman *et al.*, 2011), although the total number is unknown. Populations of sea-migrating brown trout vary in their status, from VU in the Gulf of Bothnia and the Baltic Sea to relatively stable along the west coast (Kattegat and Skagerrak) (Fiskeriverket, 2009). There has been a general improvement in sea trout parr abundance in northern and central Sweden, but a significant decline has occurred in southern Sweden. This negative trend of juvenile abundance does not necessarily reflect an overall decline in the population of sea trout in Scandinavia, based on anecdotal reports, and an increased tendency for anglers to target their fishing effort on sea trout in these areas. Data from Högvadsån (the only river with long-term data on the number of out-migrating sea trout smolts) shows a three increase in the number of out-migrating smolts between the 1960s and 2010 as a result of habitat restoration. In agreement with the overall data from southern Sweden, there was also an overall decrease in the density of 1+ parr, whereas the density of 0+ fish remained constant (Höjesjö *et al.*, 2017).

UK (England and Wales)

There are at least 70 streams and rivers supporting breeding populations of sea trout in England and Wales. Sea trout are only absent from rivers along the east coast, south of the Yorkshire Esk (54°30'N). Stocks are increasing in some previously polluted rivers (e.g. in south Wales, and northeast England). The current trends in populations are variable. The species is afforded considerable protection under the salmon legislation, but have a different, generally earlier, open season, no mandatory catch and release, and no catch limits.

UK (Northern Ireland)

Northern Ireland has 27 main salmon rivers (Crozier *et al.*, 2003), in addition to numerous small coastal streams capable of supporting sea trout. Sea trout populations are evident, to some extent, in virtually all river systems in Northern Ireland and the cross-border Foyle and Carlingford area. However some catchments (e.g. Lough Neagh) are dominated by potamodromous trout populations. Long-term electrofishing surveys on the following six rivers suggests the following changes in fry abundance since at least 2005: Bush (DCAL area) – no trend, Glendun (DCAL area) – decreasing trend, Shimna (DCAL area) – increasing trend, Roe (Loughs Agency area) – no trend, Faughan (Loughs Agency area) – no trend, Burn Dennett (Loughs Agency area) – decreasing trend.

UK (Scotland)

Sea trout occur in most rivers in Scotland. There is no standardized formal monitoring of parr. A topic sheet is produced annually which summarizes trends in reported catch. The total reported rod catch of sea trout for 2015 was 21 443. There has been a decline over much of the period since 1952. However, they appear to have stabilized in recent years and were 96% of the previous 5-year average in 2015.

2.4.8 Threats

Although not considered a threatened species, some brown trout populations do experience declining numbers as a consequence of environmental degradation in their freshwater habitat, barriers restricting their migratory routes, and exploitation. Pressures on freshwater habitats include:

- Water quality:
 - agricultural enrichment
 - pesticides and herbicides
 - sediment run-off
 - sewage treatment works
 - industrial discharges
 - aquaculture
- Water quantity:
 - loss of wetlands
 - abstraction
 - land drainage
- Habitat degradation:
 - channel modification
 - bank erosion
 - sedimentation

In many areas, estuarine and coastal habitats are also under pressure from anthropogenic developments. In addition, sea trout populations in a number of areas have been severely impacted by parasites (sea lice) from coastal salmon farming, e.g. in Norway and Ireland (Thorstad *et al.*, 2015; Thorstad and Finstad, 2018).

2.4.9 Conclusions - future

Sea trout is highly valued by both commercial and recreational fishery, and has been introduced in many countries throughout the world to increase angling opportunities. Although still numerous, sea trout populations have been affected by migration obstacles and habitat degradation in freshwater, as well as developments in estuaries (e.g. port construction) and coastal waters (e.g. salmon aquaculture). Stock monitoring is diverse, and is lacking in many regions (Section 2.4.9.1). Even catch statistics are poorly collected in some countries. In addition, there have been relatively few long-term population studies collecting smolt output and adult returns in order to establish and transport biological reference points. As a result, an accurate assessment of the stocks is lacking.

2.4.9.1 Current country-by-country sea trout stock monitoring

Denmark

Regular monitoring of trout populations is undertaken by routine electrofishing surveys in most Danish rivers ca. every 7–8 years by DTU Aqua. Ca. 7000 sites are monitored, and, from these, approximately two-thirds are electrofished. Smolt runs are irregularly monitored by traps as part of specific projects, often including PIT tagging of emigrating smolts. In the same rivers, adult returns are usually monitored at PIT reader stations.

Finland

Regular monitoring of trout populations by standardized electrofishing surveys is carried out in ten rivers holding original strains of sea trout. Moreover, several other rivers are monitored less frequently by electrofishing. Most of the data collected over the last 10 years are stored in the Finnish register of fish surveys. Smolts are regularly trapped by a modified fykenet in one large river system holding many spawning tributaries, and irregular trapping by rotating screw traps has been conducted in a couple of other rivers as part of specific projects. Catch returns are obligatory from professional sea fishers, and catch inquiries covering nationwide recreational fishery are carried out biennially. Catch samples are collected from angling in one large river system. Spawner counts are established in several fish passes and typically cover an unknown fraction of the whole spawning run. An index of spawning run size has been established in one large river system by using DIDSON sonars. Extensive tagging programmes using mostly Carlin and T-bar anchor tags have been run in conjunction with stocking activities. Finnish rivers with data collection are usually relatively wide, and monitoring is mainly targeted for Atlantic salmon (*Salmo salar*).

France

Sea trout is monitored in three trapping facilities on the Nivelle in southwest France and on the Oir and the Bresle, in lower and upper Normandy, respectively. These facilities have operated since the early 1980s and, together with detailed electrofishing surveys, provide valuable dataserries for salmonids and other migratory fish. About 25 sites with sparse dataserries are equipped with video-counting, acoustic, or fish passes at various dams or similar structures. Most counters have been installed gradually since the 2000s. Angling data are collected by the French Agency for Biodiversity (AFB), but capture declaration is not compulsory for sea trout. Thus data are scarce and do not inform on fishing pressure.

Germany

In freshwater, monitoring programmes cover summer/autumn parr abundance in stocked and some wild systems, following the THS-method proposed by WGBAST (Pedersen *et al.*, 2017). In total, about 25 Baltic Sea running rivers are monitored. In four rivers, video-camera spawner counts are conducted. For the first time in northern Germany, a spring smolt-trapping campaign was launched in 2016 and 2017 on project base. In about ten rivers, spawning beds are counted. In the Baltic Sea, two marine recreational fishing surveys were initiated by the Thünen Institute of Baltic Sea Fisheries (S. Weltersbach and H. Strehlow, in prep.). Since almost all activities in the Baltic Sea parts of Germany are only funded as projects, difficulties may arise in the continuation of monitoring activities (e.g. parr monitoring and spawning bed count, river selection). No information is available for the productivity of smolt, the returning rate of smolts, straying rates, and spatial distribution of individual populations during the marine phase. No information exists for performance differences between wild fish and individuals with stocking history (either stocked as fry or as smolts).

Ireland

Sea trout populations (typically upstream runs) are indirectly monitored in 31 of the 256 rivers where fish counters are currently or have recently been operated, in some instances complemented by trapping facilities (Table A3.2, Annex 3). The reliability of counts is variable as the equipment is primarily calibrated to monitor Atlantic salmon runs. Counters are considered to reliably detect sea trout ≥ 25 cm fork length. In addi-

tion, the time-series available vary from system to system, being dependent on the installation date of the counter and its operational efficiency within years. System-specific angling catch and commercial catch records for sea trout (> 40 cm fork length) are available for 2001–2015 from the *Salmon and Sea Trout Carcass Tagging and Logbook Scheme* (CTLS) as reported in annual reports produced by IFI and its predecessors. Reported catches are low as relatively few sea trout > 40 cm are recorded by anglers. More comprehensive, system-specific catch records (to include fish < 40 cm fork length) have been compiled based on estimates of catch supplied by regional fishery inspectors and local fishery managers.

The Erriff River system in the west of Ireland is the National Salmonid Index Catchment. Full trapping and fish counter facilities have been employed to monitor the annual sea trout population, (notably via annual returns) since 1986 and 2001, respectively. The smolt and kelt runs on the Tawnyard subcatchment are monitored via trapping facilities. Rod catch data are available since 1925. This catchment acts as an index for other sea trout populations in the west of Ireland. Counter and catch data indicate that the sea trout population in the Erriff system experiences periodic fluctuations in abundance, with no clear multiannual upward or downward trend evident in recent years. In the Burrishoole Catchment (Shramore River) in the west of Ireland, full trapping facilities have been used since 1971 to record sea trout population data, both smolt runs and adult returns, and marine survival annually. Annual rod catch data has also been recorded since that time. The abundance of smolts and adult returns remain low, with no apparent recent trend compared with the historical higher stock levels observed in the 1970s–1980s (Poole *et al.*, 2006; MI, 2016). Extensive fish population monitoring takes place on river systems throughout Ireland that contain sea trout via a range of programmes (e.g. for the EU Water Framework Directive and Salmon Catchment-wide Electrofishing Programme). However, juvenile sea trout-specific population data have to be reported as total juvenile trout, because this life stage cannot be distinguished from cohabiting juvenile resident brown trout populations.

Norway

The only national register related to the abundance of sea trout is the collection of catch reports from freshwater recreational fishery (see Statistics Norway (www.ssb.no); Anon., 2015). However, these catches are generally underreported, especially in smaller watercourses with poor administration of the licensed fishery. Catch returns are obligatory from professional sea and freshwater fishers. A new national register for biological and environmental data collected during environmental assessments, including densities of juvenile sea trout, is under construction¹³. However, only limited information on sea trout is available to date. Several institutions (e.g. NTNU University Museum, Norwegian Institute of Nature Research, Uni Research) undertake regular surveys to assess 0+ and older parr densities by electrofishing from late summer to early autumn, and visual counts are made of the numbers of spawning fish in many salmon rivers by observations on foot, from boats, or by drifting divers. Smolt runs are monitored with Wolf traps in two index rivers, or occasionally (typically for 1–5 years) by rotary screw traps, PIT tag antennae, submerged video cameras, or other temporarily installations. Returning adults are counted in Wolf traps in the index rivers or, occasionally (typically for 1–5 years), by PIT tag antennas, submerged video cameras, or other temporary installations.

¹³ <http://vannmiljo.miljodirektoratet.no>

Poland

Sea trout is monitored on a project-by-project basis, both for fishery and conservation, and targets both the pre-adult and adult stages. An inventory of sea trout spawning redds has been done in 3–5 rivers annually since 2003. Monitoring of parr densities has been carried out since 2004 in eight rivers in northern Poland at 1–12 electrofishing sites per river. There are some fish counters in fish passes. Data from two of them, in the Slupia and Vistula rivers, are credible and can be considered as population monitoring data (the first counter has been working since 2006, the second since 2015). Data on stocking, commercial catches, and other activities in rivers are gathered by the Inland Fisheries Institute and provided to ICES.

Portugal

Despite the fact that the Minho sea trout population is probably the largest in Portugal, studies about its bioecology are almost inexistent. The same applies to the majority of river basins in which the species occurs. The population inhabiting the River Lima is the only one in Portugal for which some bioecological information is available (Maia, 2003). According to this study, upstream spawning migration of sea trout adults occurs between November and June and can be divided into two migration periods: (i) autumn and winter, in which the majority of adult sea trout migrate upstream to spawn; and (ii) spring, which seems to be less important than the first period, with only a few trout showing this behaviour. Tagged adult sea trout (i.e. radiotelemetry) moved distances up to ca. 8500 m upstream into one of the main tributaries of River Lima (River Estorãos). Smolts were observed migrating downstream between February and May. In this study, distinct methods (i.e. scale reading and identification of phenotypic expression of smoltification) were used to identify the proportion of smolts within trout populations occurring in River Lima and studied tributaries (Estorão, Vez, and Vade). Smoltification rates in the study area varied between 13 and 25%. Within the studied sea trout population in the Lima river basin, four different smoltification ages were identified, namely 1+, 2+, 3+, and 4+. Smolts were predominantly from the age-classes 1+ (31.3%) and 2+ (54.5%), meaning that most juveniles spend one or two years in freshwater before their first migration to the estuary and/or sea. Within the Lima population, two strategies were identified for sea trout, regarding the residence time of sea trout in marine and/or estuarine environments: (i) trout that spent less than one year in marine/estuarine environments; and (ii) trout that spent at least one year in marine and/or estuarine environments before migrating into freshwater to spawn.

Spain

The only CCAAs with some degree of information available about sea trout abundance trend are Navarre and Galicia. In Navarre, the only river with sea trout is the Bidasoa. In this river, a regular upstream trap has been obtaining information about the adult sea trout run since 1995 (Figure A3.7, Annex 3). In addition, at one sample point of this river, electrofishing surveys have been done to assess the trend in juvenile density since 1992 (Figure A3.8, Annex 3). In Galicia, three regular upstream traps have operated since the 1990s in the rivers Ulla, Lérez, and Tea (Minho Tributary) (Figure A3.7, Annex 3). Two of these have smolt traps (Ulla and Tea River), although no estimation of smolt natural production has yet been obtained. Since 1995, electrofishing surveys have been done to assess brown trout abundance, including sea trout waters (Figure A3.7, Annex 3). Average juvenile trout densities for the period of study are relatively low in the sea trout areas from Galician and Bidasoa rivers (Galicia Atlantic rivers: 10.9 100 m⁻², Ga-

Galician Cantabrian rivers: 10.2 100 m⁻², and Bidasoa River: 3.4 100 m⁻²). The trend in juvenile density in these rivers in this period is more or less stable, and no correlation has been found between the rivers studied (Figure A3.7, Annex 3).

Since 1992, anglers have been required by law to declare their catches; and, accordingly, an official series of sea trout catches has been produced since 1995 (Figure A3.9, Annex 3). In the three Galician upstream traps, the trend in Galician sea trout run in the period analysed is negative, but in the Bidasoa trap, the trend is positive (although perhaps in the first years of the study period, sea trout were underestimated). The river where most sea trout are caught at traps is the Tea (about 985 individuals per annum), markedly greater than in Ulla (266), Lérez (207), or Bidasoa (52) traps (Figure A3.8, Annex 3). Changes in size limit have distorted the Galician sea trout catch series (Figure A3.9, Annex 3). Catches dropped from 3000 sea trout declared in 1995 to around 700 in 2011–2013. When analysing this series without the size limit change effect, the trend is also negative, as in the Galician traps series. One reason for the autumn in catch was a continuous pyrite discharge into the Eume River (the river with higher sea trout official catch in Galicia, see red colour in Figure A3.8, Annex 3) which led to the collapse of this sea trout population in 2007–2008.

UK (England and Wales)

Monitoring is undertaken for both conservation and fishery reasons. Parr abundance is monitored by annual electrofishing surveys (800–1000 sites), and smolts and adults are monitored by trapping in five rivers. Catch statistics are collected for all net and rod fishery.

UK (Northern Ireland)

The main monitoring is via semi-quantitative electrofishing fry surveys (Crozier and Kennedy, 1995). Annual surveys have been undertaken on six rivers since at least 2005. In addition to the fry monitoring programmes, two index rivers for sea trout are being established. In the Loughs Agency cross-border Foyle and Carlingford catchment area, the Burn Dennett is being developed as an index catchment for sea trout with increased monitoring of production and adult returns. In the DCAL area of Northern Ireland, the Shimna is being developed as an index river for sea trout.

UK (Scotland)

No standardized monitoring programme exists, although electrofishing surveys are conducted on many Scottish rivers, and there is a legal obligation on all net and rod fishery to report fishery statistics¹⁴.

2.4.9.2 Recommendations

Fundamental to the management of sea trout is the development of Biological Reference Points (BRP) and/or alternative measures for stock assessment. Such scientifically-based population models will provide an enhanced basis to inform the management of stocks (ICES, 2013b). This process has progressed substantially for salmon populations, but has been considerably slower for sea trout, a fact that has been recognized at the international level (ICES, 2013b). Over the last two decades, considerable work has been done on salmon stocks in Ireland, and elsewhere, to inform the development of BRPs (e.g. conservation limits; SSCS, 2016). A similar focus is required for sea trout.

¹⁴ <http://www.gov.scot/Topics/marine/Publications/stats/SalmonSeaTroutCatches>

The basis of this work has commenced in Ireland (e.g. in the Erriff system, which is the National Salmonid Index Catchment), but further development on more regional and national levels is desirable.

Establishing valid stock–recruitment relationships for sea trout populations may be more difficult than for salmon. This is resulting from both the large number of trout rivers entailed, and to the greater complexity and variability of the sea trout's life cycle, with a proportion of the population being resident. Hence, stock–recruitment relationships will only be applicable for a limited number of rivers that possess similar characteristics, or are located in the same geographic region. Therefore, it will be important to establish representative electrofishing sites in different regions. Many countries already have long-term data from sites that have been fished regularly, even if the data on smolt productivity is often lacking. It is suggested that a project should be initiated to select a larger number of index rivers along a north–south gradient, and monitor data on juvenile abundance (electrofishing) and productivity (e.g. out-migrating smolts or similar approaches). Although time-consuming to establish, it will allow the development of stock–recruit relationships. In 2017, ICES Working Group WGTRUTTA was established, with one of its aims being to identify candidate sea trout index rivers for monitoring and assessment.

In parallel, it is also recommended to further develop the work initiated within the Baltic to assess smolt production using a combination of electrofishing and habitat surveys. This method is based on the standardized procedure of quantifying trout habitat, and combining this information with modelled estimates of smolt production, using the mean density of > 0+ trout parr (pre-smolt) in autumn (Höjesjö *et al.*, 2017). It is important to both evaluate and validate this approach in different regions, and to link this work to the project of establishing index rivers across regions. WGTRUTTA aims to develop BRPs of stock indices based on catch or juvenile data, in combination with habitat surveys. This model could also be added as a tool to aid restoration and prioritization of activities when restoring degraded habitat, particularly in smaller streams that are important for sea trout production (Whelan, 2014; IFL, unpublished). In the case of habitat restoration, additional measures may also need to be explored to enhance the resilience of habitats to natural pressures, such as washout of spawning and nursery areas from flooding events..

Furthermore, more systematic recording of catch in recreational fishery, which includes fish < 40 cm fork length, is desirable, in order to provide more reliable and meaningful catch data for different types of assessments. However, it must be emphasized to bear in mind that such statistics will include stocked fish as well as fish migration from different areas. In order to obtain a proper overview of wild sea trout abundance, it will be necessary to always estimate the abundance of naturally spawned juveniles, although, in some systems, most juveniles will become resident trout.

It is also recommended that exploitation be reduced, especially in the Bothnian sea (ICES subdivisions 30 and 31) and that construction of new barriers be avoided. Fishing pressure should also be reduced in the southeast Baltic (ICES Subdivision 26) and in the south (ICES subdivisions 22 and 24). Habitat improvements are needed in many places, and accessibility to and from spawning areas should be greatly improved (ICES, 2016b).

2.5 Sturgeon

The following sections summarize information on the sturgeon species that are native to the European ICES range and diadromous, utilizing both freshwater and coastal habitats. In Europe, anadromous sturgeons (*Acipenser* spp.) originally inhabited the continental shelf waters of the Northeast Atlantic Ocean and the Mediterranean and Black seas. While the species persisted in the Mediterranean and Black seas as well as on the Iberian Peninsula, most of the northern European Atlantic coastal waters including the North and Baltic seas, were colonized ca. 4500–3500 years BP (Ludwig et al., 2008).

With regard to management implications, it is suggested that the species from one river should be considered as distinct management entities from species spawning in other rivers, even when the genetic differences are insignificant, owing to the adaptation to environmental conditions and homing behaviour. Evidence for low straying rates (< 4% of the population) was determined in the Gulf sturgeon (*A. oxyrinchus desotoi*) by Stabile *et al.* (1996), and *A. sturio* colonization along the English Channel and into the North Sea evidently took centuries (Nikulina and Schmölcke, 2017).

2.5.1 Adriatic sturgeon

2.5.1.1 Taxonomy

Class:	Actinopterygii
Order:	Acipenseriformes
Family:	Acipenseridae
Scientific name:	<i>Acipenser naccarii</i>

Subspecies, variations, synonyms

None

Taxonomic notes

None

Common name

AL: Blini i Adriatikut, HR: Jadranska jesetra, DE: Adriastör, EN: Adriatic sturgeon, FR: Esturgeon de l'Adriatique

General characteristics

Maximum length:	220 cm
Maximum weight:	120 kg

The Adriatic sturgeon (*Acipenser naccarii*) is endemic to the northern part of the Adriatic Sea and its tributaries (Figure 2.12). In the Po River, the species was less common than *Acipenser sturio* until the 1970s, and it has followed the decline of all Italian sturgeon species. It has been observed in all the larger rivers in the northern Adriatic (Adige, Brenta, Bacchiglione, Livenza, Piave, Tagliamento, and Sile), and from the Drin and Buna rivers, and Skodra Lake on the Albanian–Montenegro border. It is very occasionally reported along the Greek coast from the Ionian Sea to Corfu.



Figure 2.12. Distribution of *Acipenser naccarii* (http://www.ittiofauna.org/webmuseum/pesciossei/-acipenseriformes/acipenseridae/acipenser/acipenser_naccarii/index.htm, accessed 11.12.2017).

2.5.1.2 Life cycle and migrations

Maturity is reached after six years in males and eight years in females. The species is mostly limited to freshwater and brackish water habitats (potamodromous). It is generally not found in marine waters, but it does occur along the Adriatic east coast until Corfu (Tortonese, 1989). The salinity tolerance of juveniles of up to two years of age is limited to ca. 20 ppt during acute exposure (Cataldi *et al.*, 1999), but this tolerance increases with age.

Reproduction occurs from May to July. The upstream spawning migration into Italian rivers occurs during the first months of the year (D'Ancona, 1924; Paccagnella, 1948).

2.5.1.3 Stock structure and population dynamics

In the early 2000s, there was considerable debate around the status of *A. naccarii* in Spain, but no proof for the historical presence of the species has been provided (Almacá and Elvira, 2000; Garrido Ramos *et al.*, 1997; Ludwig *et al.*, 2003). In Greece, the species was described historically, but is currently considered missing (Economides, 1973). A restocking attempt was carried out at the beginning of the 2000s without any effect described (Patschos, 2003).

2.5.1.4 Genetics

The genetic population structure of the broodstock in Italy is well documented (Congiu *et al.*, 2011). The data are currently used to identify wild and restocked fish when captured. Differences between Albanian and Italian populations have been assessed (Ludwig *et al.*, 2003), and resulted in the proclamation of separate Distinct Population Segments (DPS).

2.5.1.5 International status with country highlights

International

OSPAR Convention:	Not listed
EC Habitats Directive:	Annexes II and IV (92/43/EEC)
CITES:	Appendix II
Bern Convention:	Appendix II
Bonn Convention:	Appendix II
IUCN Criteria:	Subcriteria A2abc, B2ab (i, ii, iii, iv, v)
Global IUCN Red List Category:	CR (2010)
HELCOM Red List:	Not listed

National

Currently, it must be considered CR in Italy (Table 2.10). A captive broodstock with controlled reproduction was established in 1988 (Arlati *et al.*, 1988). This has allowed recovery stocking since the 1990s (Arlati *et al.*, 1999, 2003). The species is reported as either vagrant or native in Croatia, Macedonia, Albania, and Greece.

Table 2.10. National and IUCN Red List classifications for the Adriatic sturgeon. (Countries in which the species is not listed - out of species range: Belgium, Bosnia-Herzegovina, Canada, Czech Republic, Denmark, Estonia, Finland, France, Germany, Iceland, Ireland, Latvia, Lithuania, The Netherlands, Norway, Poland, Portugal, Russia, Sweden, UK (England and Wales, Northern Ireland, Scotland), and USA).

Country	Spawning populations	Status	Remarks	References
Albania	Yes	EN	No criteria mentioned	National Red List 2013
Croatia	Yes	CR/EX		Mrakovčić <i>et al.</i> (2006)
Greece				Barbieri <i>et al.</i> (2015)
Italy	Yes	CR	A2c; C2a(ii); D	Rondinini <i>et al.</i> (2013)
Montenegro	Yes	EX	Drin River	IUCN (2010)
Slovenia			Status unclear	
Spain	No	NA	Not listed, out of species range	Almaca and Elvira (2000)

DD: data deficient; VU: Vulnerable; NT: Near Threatened; EN: endangered; CR: Critically endangered; RE: regionally extinct; EX: extinct; NA: not applicable

2.5.1.6 Distribution summary

The species is very rare in the wild. Most of the captures in the Po River and the northern Adriatic Sea originate in releases in Italy. One self-sustaining population is postulated in the River Ticino. More recently, several big adult sturgeons have been caught in the recreational fishery for the exotic European catfish (*Silurus glanis*) in the Po River, some of which were genetically allocated to populations restocked in the past (Bronzi, pers. comm.).

2.5.1.7 Threats

The species has undergone a massive decline mostly because of overfishing (Bronzi *et al.*, 1994) and habitat loss. Habitat degradation during the last decades has mainly resulted from continued environmental pollution, and the construction of artificial dams and weirs along almost all rivers that the species formerly inhabited (Cataldi *et al.*, 1995). Some progress is being made. For example, a recently constructed fish passage at the Isola Serafini Dam, on the middle of the Po River, should permit the migratory movements of this species, opening the river to migratory movements into the upstream part of the river for downstream populations (Life Conflupo project¹⁵).

2.5.1.8 Conclusion – future

Recommendations:

- A status assessment should be carried out for the Adriatic sturgeon in Italian waters to determine the fate of the species with regard to its natural reproduction and recruitment. The potential for natural recruitment exists, owing to the fact that sexually mature fish are probably present in the river .
- The role of the dominant exotic fish community in the Po River and its impact on the native fish fauna should be investigated.
- The status of the species in Albania has not been monitored since 1995 and should be updated.

¹⁵ <http://www.life-conflupo.eu/prj2013/index.php?lang=en>

2.5.2 Baltic sturgeon

2.5.2.1 Taxonomy

Class:	Actinopterygii
Order:	Acipenseriformes
Family:	Acipenseridae
Scientific name:	<i>Acipenser oxyrinchus</i>

Subspecies, variations, synonyms

None

Taxonomic notes

None

Common name

EN: American Atlantic sturgeon, Baltic sturgeon, DK: Vestatlantisk stør FI: Sinisampi, FR: Esturgeon Noir, PL: Jesiotr ostronosy, SE: Stör

General characteristics

Baltic sturgeon is an anadromous species that reproduces in freshwater and spends most of its life at sea. In the 19th century the Baltic sturgeon was widespread, but it reproduced only in rivers entering the southern part of the Baltic Sea and lake Ladoga. While *A. sturio* (section 2.5.3) and *A. oxyrinchus* have been found in excavation material in the northern range, more recent samples show a clear segregation between the Baltic Sea (*A. oxyrinchus*) and the North Sea, and Northeast Atlantic (*A. sturio*). The species entered most large rivers draining into these marine waters (Magnin, 1959; Freyhof, 2002). *A. oxyrinchus* is considered extinct or missing in the Baltic Sea, whereas catches of single individuals in UK (2006) and Spain (2010) indicate some unidentified population remain.

In North America, Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) can be still found along the entire western Atlantic coast from Labrador, Canada, to Florida. One subspecies, *A. oxyrinchus desotoi*, ranges from Florida to Mississippi in the Gulf of Mexico. The species is anadromous, migrating from the ocean into coastal estuaries and rivers to spawn.

Recent research has shown that the *A. oxyrinchus* was present along the Atlantic coast of Europe at least until 1500 BP. Baltic Sea populations existed until the early 20th century, when overharvest, hydro construction, and pollution drove them to extinction. Because of the historical presence of *A. oxyrinchus* in the Baltic Sea, the reintroduction of this species is justified and in line with legal guidelines. Reintroduction measures are underway in the countries along the southern Baltic Sea, namely Germany, Poland, Lithuania, Latvia, and Estonia, using genetically similar broodstock and offspring imported from the Saint John River, NB, Canada. Captive broodstocks (*ex situ*) exist in Germany and Poland and comprise several hundred individuals. Releases have been carried out in the rivers Oder and Vistula since 2006, and in the rivers Pregolia, Nemunas, Daugava, and Venta since 2011. Releases are mostly conducted with feeding fry, fingerlings of 1–3 month of age, and yearlings. The total number of fish released since 2006 has reached 3 million individuals of different ages. Monitoring includes migration analysis, habitat use (Fredrich et al., 2008), and bycatch analysis in commercial fishery based on deliberate reports following communication campaigns (Gessner et al., 2010).

Recapture data show extensive migration of the fish into the North Sea, English Channel, and Bay of Biscay as well as all across the Baltic Sea. The first evidence for homing into the Oder River was obtained in 2016, when an adult sturgeon was caught illegally by an angler. Harmonization of the reintroduction measures is subject to a HELCOM project group.

2.5.2.2 Life cycle and migrations

Acipenser oxyrinchus show wide variations in population characteristics throughout their range. Growth, age at first maturation, spawning frequency, and longevity largely depend on latitude. Maximum age ranges from 45 to 120 years. Females reach sexual maturity between the ages of 7 and 30 years, and males between the ages of 5 and 24 years.

Acipenser oxyrinchus is an anadromous species. Reproduction generally occurs in the lower parts of rivers; and larvae and juveniles progressively drift downstream. The species has a limited ability to osmoregulate in hyperosmotic conditions during the early life phases. This is a potential cause for its restriction to the low salinity habitats of the Baltic Sea. Growout in juveniles and subadults mainly occurs in coastal waters of up to 40 m water depth. Larger specimens (> 80 cm) show an increased ability to osmoregulate and utilize habitats in the western Baltic and the Kattegatt area (Gessner, unpubl.).

In estuaries, *A. oxyrinchus* feeds on benthic invertebrates (crustaceans, Oligochaeta, Polychaeta). As it increases in size, it also feeds seasonally on small benthic fish (Guilbard *et al.*, 2007).

2.5.2.3 Stock structure and population dynamics

Population estimates are available for North American rivers (ASSRT, 2007). In Europe, the species is considered lost or extinct, with restoration programmes ongoing. No population assessments have been carried out yet for any of the populations under restoration in the Baltic.

2.5.2.4 Genetics

The wild populations of *A. oxyrinchus* comprise at least seven distinct population segments (DPS) along the North American Atlantic seaboard. Baltic sturgeons have been shown to be genetically similar to haplotype A from the northern DPS (Ludwig *et al.*, 2002, 2008; Popovic *et al.*, 2014). This haplotype is dominant in the rivers between Connecticut and Quebec. The stocking material and broodstock for the restoration of the Baltic Sea originates from this DPS, specifically from the commercial fishery in the Saint John River (Gessner *et al.*, 2010).

2.5.2.5 International status with country highlights

International

OSPAR Convention:	Not listed
EC Habitats Directive:	Not explicitly listed, but considered as included under Annex II and V (it was not known to be a separate species in the Baltic when the Directive was compiled)
CITES:	Appendix 2

Bern Convention:	The Baltic population of the Atlantic sturgeon should be considered included under Annex III on similar grounds as for the Habitats Directive
Bonn Convention:	Not listed
IUCN Criteria:	-
Global IUCN Red List Category:	NT for the North American range, with a population decline of 30% over the last 10 years, or > 50% in the long term, NE for Baltic population.
European IUCN Red List Category:	NE
HELCOM Red List:	RE (HELCOM, 2007). A restoration action plan is in the final stages of approval by HELCOM for the Baltic range states.

National

Although "Not Listed" by the US Fish and Wildlife Service, the following DPSs are listed by the National Marine Fishes Service (2010) as of February 2012 as ESA Endangered: New York Bight DPS, Chesapeake Bay DPS, Carolina DPS, South Atlantic DPS; or ESA Threatened: Gulf of Maine DPS.

The species is protected nationally in Denmark, Estonia, Germany, Latvia, Poland, and Sweden. A complete list of national classifications can be seen in table 2.11.

Table 2.11. National and IUCN Red List classifications for *Acipenser oxyrinchus*.

Country	Spawning populations	Status	Remarks	References
Albania	No	NA	Not listed, out of species range	National Red List of 2013
Belgium	No	NA	Not listed, out of species range	Verreycken <i>et al.</i> (2014)
Canada	Yes	VU		COSEWIC (2011)
Czech Republic	No	NA	Not listed, out of species range	Lusk <i>et al.</i> (2015)
Croatia	No	NA	Not listed, out of species range	Mrakovčić <i>et al.</i> (2006)
Denmark	No	EX	Coastal catches only, not listed in Danish Red List	Carl and Möller (2012)
Estonia	Yes	CR/EX	Listed as <i>A. sturio</i>	Estonian Red List (2013)
Finland	Unclear	RE		Rassi <i>et al.</i> (2010)
France	No	NA	Not listed, out of species range	IUCN France <i>et al.</i> (2010)
Germany	Yes	EX or missing	Last catch in the Oder River 1968	Haupt <i>et al.</i> (2009)
Greece	No	NA	Not listed, out of species range	Barbieri <i>et al.</i> (2015)

Table 2.11 (continued)

Country	Spawning populations	Status	Remarks	References
Iceland	No	NA	No records in inland waters, only single individuals caught along the shores	
Ireland	No		No confirmation of former spawning populations	
Italy	No	NA	Not listed, out of species range	
Latvia	Yes	EX	Listed as <i>A. sturio</i>	Kottelat and Freyhof (2007); Medne (pers. comm.)
Lithuania	Yes	EX	Listed as <i>A. sturio</i>	Bukantis <i>et al.</i> (2014)
Netherlands	No	NA	Not listed, out of species range	
Norway	No	NA	Not listed, out of species range	
Poland	Yes	EX	Listed since 2016 (Wolos, pers. comm.)	Glowacinski <i>et al.</i> (2002)
Portugal	No	NA	Not listed, out of species range	
Russia	No	EX	Entered and spawned in the Neva River system and in the Pregola (Kaliningrad), species status not yet solved	Popov (2017)
Spain	No	NA	Not listed, out of species range	Almacá and Elvira (2000)
Sweden	No	NA	Never reproduced in Sweden. New information says that the <i>Acipenser</i> species that reproduced in Sweden more than 100 years ago was <i>A. oxyrinchus</i>	Gärdenfors (2005)

Table 2.11 (continued)				
Country	Spawning populations	Status	Remarks	References
UK (England and Wales)	Unclear	EX	Status unclear, Reports of occurrence from the Severn River unconfirmed	
UK (Northern Ireland)	No	NA	No spawning rivers known	
UK (Scotland)	No	NA	Status unclear	Parnell (1838)
USA	Yes	NT		ASSRT (2007)

DD: data deficient; VU: Vulnerable; NT: Near Threatened; EN: endangered; CR: Critically endangered; RE: regionally extinct; EX: extinct; NA: not applicable

2.5.2.6 Distribution summary

In North America, the species ranges from Quebec (Canada) to Florida (USA), with varying conservation status (Figure 2.13). In the Baltic Sea, the species is considered extinct or missing. Historic records state that at the turn of the 18th century, the Baltic sturgeon reproduced in the rivers entering the southern Baltic: Oder and tributaries, Vistula and tributaries, Pregola, Nemunas, Venta, Gauja, Salaca, Daugava, Irbe, Narva, and Neva rivers (Figure 2.14). A landlocked population was described from Lake Ladoga, with spawning taking place in the Volchov River (Berg, 1935; Popov, 2017).

Incidental captures of stocked fish are reported along the coasts of the Baltic states from the Bothnian Bay to the Kattegat, and rarely from the North Sea, the English Channel, and the Bay of Biscay. Catch reports account for more than 2500 individuals (Arndt, pers. comm.).

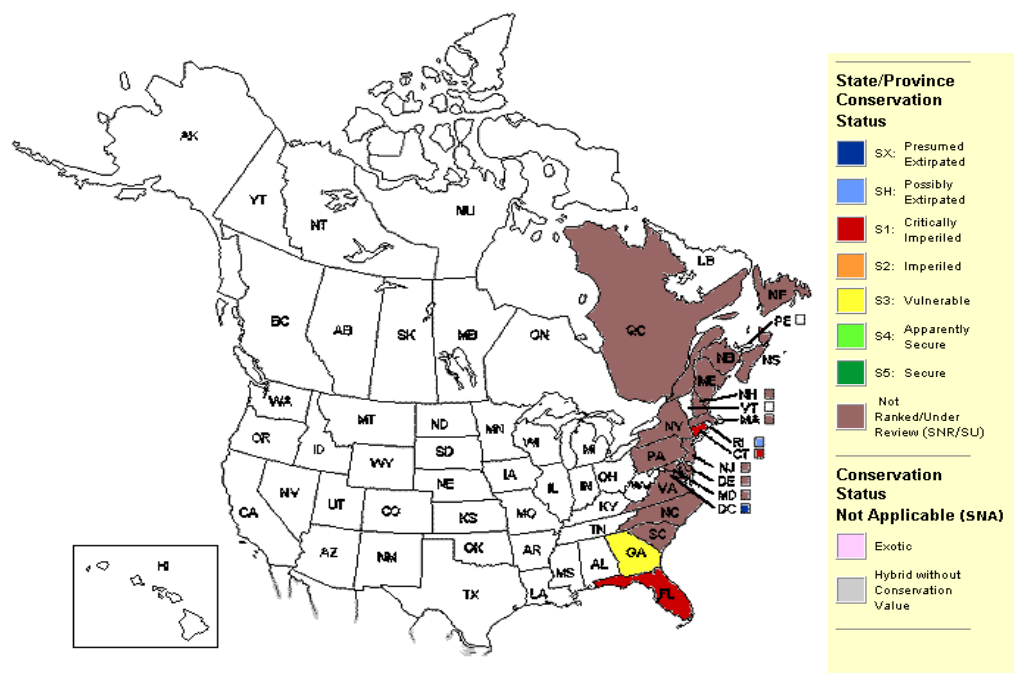


Figure 2.13. Range and conservation status of *A. oxyrinchus oxyrinchus* in North America (<http://explorer.natureserve.org/servlet/NatureServe?searchName=ACIPENSER+OXYRINCHUS>).



Figure 2.14. Spawning rivers of *Acipenser oxyrinchus* in the Baltic Sea catchment (blue) and sections utilized for spawning until 1900 (green); the populations became dysfunctional after 1950. Last catches are indicated by the date giving the year and place of catch.

2.5.2.7 Threats

During Medieval times, *Acipenser oxyrinchus* supported commercial fisheries of varying magnitude. Landings in the Baltic peaked in the 17th and 18th centuries (Debus, 1995), whereas landings in the US reached their maximum just prior to 1900, with an estimated 7 million pounds per year (Secor, 2002). Commercial fishery has massively contributed to the population decline of this species. Overfishing was already apparent in the 18th century, and has been the main cause for the extinction of sturgeon populations during the last two centuries (Debus, 1995). Pollution, river regulation, and hydro construction further accelerated population declines (Mamcarcz, 2000; Gessner et al., 2010). Based on modelling efforts, Jaric and Gessner (2013) determined that the safe limit for removals from the population is 1–4% annually, in order to avoid a strong decline in population size and an impairment of age structure.

Dams and weirs in rivers can massively affect the spawning migration. Habitat loss attributable to dam construction and water pollution are thought to be major factors impeding the full recovery of populations (Gilbert, 1989; Secor and Gunderson, 1997).

Degraded water quality could affect the early life survival or development of sturgeons (Delage, 2015). In Chesapeake Bay, and elsewhere in the range, hypoxic events have increased and are thought to degrade nursery habitat for Atlantic sturgeon (Secor and Gunderson, 1997).

Climate change can alter the overall suitability of formerly frequented habitats, increasing uncertainty on where to reintroduce the species (Lassalle et al., 2010). The introduction of alien sturgeon species could cause disease transmission and competition (Arndt et al., 2000), as well as confusion between protected and alien species in case of incidental captures.

2.5.2.8 Conclusion – future

The four main *Acipenser ocyrinchus* DPS along the American eastern seaboard have been strongly impacted by overharvesting for flesh and eggs, pollution and hydro construction. Populations in Canada are still commercially exploited. Due to the continuous failure of attempts to recover the populations, the US states and federal government implemented a coastal-wide moratorium in late 1997 and early 1998. This moratorium called for the rebuilding of 20 year classes, which it was estimated would take 20–40 years on from 1998. In the Baltic Sea, range reintroduction measures are underway in Estonia, Germany, Lithuania, Latvia, Poland, and Russian Federation (Kalinin-grad).

To ensure the viability of the sustained populations, long-term and coordinated recovery actions need to be implemented. Recovery will require 30+ years and intensive international cooperation to ensure survival of the species at sea, owing to its extensive marine migrations, late maturation and longevity.

2.5.3 European sturgeon

2.5.3.1 Taxonomy

Class:	Actinopterygii
Order:	Acipenseriformes
Family:	Acipenseridae
Scientific name:	<i>Acipenser sturio</i>

Subspecies, variations, synonyms

None

Taxonomic notes

None

Common name

AL: Blini; EN: European sturgeon, FR: Gaizcata, Sturione, Esturgeon commun, ES: Esturión

General characteristics

In Europe, anadromous sturgeons (*Acipenser* spp.) historically inhabited the continental shelf waters of the Northeast Atlantic Ocean and the Mediterranean Sea. They entered large rivers draining into these waters (Magnin, 1959; Freyhof, 2002). Since 1970, sturgeons have only rarely been caught in their historical range. *Acipenser sturio* is now reduced to a relict population that originated in the French Gironde River. A captive stock composed of wild spawners was built in France in the 1990s (Williot et al., 2007). Since 2007, stocking has occurred in the Gironde in France (MEDDTL, 2011) and in the Elbe in Germany (Gessner et al., 2010).

2.5.3.2 Life cycle and migrations

Acipenser sturio is an anadromous species. Reproduction generally occurs in the lower parts of rivers, larvae and juveniles progressively drift downstream, and juvenile growth takes place mainly in estuaries and at sea (Acolas et al., 2011). Reproduction occurs in May–June (Magnin, 1962). Males are mature at 10–12 years old, and females at 13–16 years old. Spawning ground characteristics were described in the Garonne and Dordogne watershed by Jégo et al. (2002): a deep pool including a sector above 5 m depth, heterogeneous substrate (3–250 mm), and water current of 0.5–1.5 m s⁻¹.

Juveniles are assumed to leave the river (freshwater) for the estuary (low salinity sectors) before age one (Rochard et al., 2001; Acolas et al., 2017). In the estuary, they feed on polychaeta and some crustaceans (Ninua, 1976; Brosse et al., 2000). At sea, incidental capture reports have shown that wild individuals are found mainly between 10 and 40 m depth (maximum 100 m), with sizes currently ranging between 35 and 244 cm (Letaconnoux, 1961; Rochard et al., 1997).

2.5.3.3 Stock structure and population dynamics

No population estimates are available. Wild-origin individuals are very scarce, and the last natural reproduction was reported in the Gironde watershed in 1994 (Lochet et al., 2004, Williot et al., 2007). Currently, incidental captures of stocked fish are reported along the coasts between northern Spain and the North Sea (> 1000 individuals, Acolas, pers. comm.), but fish are not yet mature.

2.5.3.4 Genetics

The relictual wild population of *A. sturio* originated from the Atlantic coast populations, which was characterized by less variability than the southern populations (Chassaing *et al.*, 2016). This depletion in the original genetic diversity (Chassaing, 2010) requires challenging conservation strategies. Genetic erosion has been taken into account during the recent stocking programmes by notably promoting breeding between unrelated mates, and carrying out a careful genetic monitoring (Roques *et al.*, 2018).

2.5.3.5 International status with country highlights

International

OSPAR Convention:	Of global and local importance, rare, sensitive, and in decline
EC Habitats Directive:	Annexes II and IV
CITES:	Appendix I, international trade banned
Bern Convention:	Annex III
Bonn Convention:	Appendix I and II
IUCN Criteria:	A2d subclassification - there has been an observed, inferred, or suspected population size reduction of $\geq 80\%$ over the last 10 years or three generations, where the reduction or its causes may not have ceased. Cause indicated - potential levels of exploitation
Global IUCN Red List Category:	CR
European IUCN Red List Category:	CR
HELCOM Red List:	EX

National

A fishing ban was implemented in 1982 in France. A complete list of national classifications for European sturgeon can be seen in Table 2.12.

Table 2.12. National and IUCN Red List classifications for *Acipenser sturio*.

Country	Spawning populations	Status	Remarks	References
Albania	Yes	EN	Buna River system	National Red List 2013
Belgium	Yes	RE	Flanders region	Verreycken <i>et al.</i> (2014)
Canada		NA	Out of the known species range	
Czech Republic	Yes	RE	Elbe tributaries	Lusk <i>et al.</i> (2015)
Croatia	No	RE	Individuals in the Neretva River	Mrakovčić <i>et al.</i> (2006)

Table 2.12 (continued)

Country	Spawning populations	Status	Remarks	References
Denmark	No	EX	Individuals in the Gudenå River	Wind and Pihl (2004)
Estonia	Yes	EX	listed instead of <i>A. oxyrinchus</i>	Estonian Red List (2013); Tambets (pers. comm.)
Finland	unclear	EX	<i>A. oxyrinchus</i> listed, individuals in the Kemijoki River (Gösta <i>et al.</i> , 1883)	Rassi <i>et al.</i> (2010)
France	Yes	CR	Last remaining population	IUCN France <i>et al.</i> (2010)
Germany	Yes	CR	Last catch in the Elbe River in 1993	Haupt <i>et al.</i> (2009)
Greece	Yes	DD	Past occurrences in the Evros and Struma rivers, current and past situations unclear	Barbieri <i>et al.</i> (2015)
Iceland	No	NA	No records in inland waters, only single individuals caught along the shores	
Ireland	No		No confirmation of former spawning populations	King <i>et al.</i> (2010)
Italy	Yes	EX	Last specimens caught in 1994 in the Po River	Elliot and Hemingway (2002)
Latvia	Yes	EX	listed instead of <i>A. oxyrinchus</i>	Kottelat and Freyhof (2007); Medne (pers. comm.)
Lithuania	Yes	EX	Former spawning population in the Neman River, restoration plan in place since 2011	Lithuanian Red List (2007)
Netherlands	Yes	EX		de Nie (2003)
Norway	No	NA	Occasional marine catches	
Poland	No	NA	see <i>A. oxyrinchus</i>	Głowacinski <i>et al.</i> (2002); Wolos (pers. comm.)
Portugal	Yes	RE		Almaca and Elvira (2000)

Table 2.12 (continued)

Country	Spawning populations	Status	Remarks	References
Russian Federation	DD	EX	Sturgeons entered and spawned in the Neva River system, but uncertainties on whether the species is <i>A. oxyrinchus</i> or <i>A. sturio</i>	Barannikova and Holcik (2000)
Spain	Yes	EX	Last catch in the Guadalquivir in 1992	Almacá and Elvira (2000)
Sweden	No	NA	Never reproduced in Sweden. New information says that the <i>Acipenser</i> species that reproduced in Sweden more than 100 years ago was <i>A. oxyrinchus</i>	Gärdenfors (2005)
UK (England and Wales)	unclear	EX	No reproduction confirmed for the English North Sea tributaries; Don catchment, last catch in 1871. Other rivers on the North Sea coast (Humber, Thames) likely similar	Howes (1997); Holcik <i>et al.</i> (1989)
UK (Northern Ireland)	No	NA	No spawning rivers known	
UK (Scotland)	No	NA	Status unclear	Parnell (1838)
USA	No	NA	Outside species range	

DD: data deficient; EN: endangered; EX: extinct; NA: not applicable; RE: regionally extinct

2.5.3.6 Distribution summary

At the turn of the 18th century, the European sturgeon had an almost pan-European distribution (Figure 2.15). Twenty-four large basins held self-sustaining populations in 1850, such as the Rioni (Georgia; 20), Danube (Romania/Ukraine; 4), Ebro (Spain; 6), Guadalquivir (Spain; 11), Guadiana (Spain/Portugal; 12), Gironde–Garonne–Dordogne (France; 10), Seine (France; 21), Rhine (The Netherlands; 18), Ems (Germany; 9), Weser (Germany; 24), Elbe (Germany; 8), Eider (Germany; 7) (Magnin, 1959; de Groot, 2002; Lassalle *et al.*, 2010, 2011). This number had decreased to 18 in 1950 (Figure 2.16). In the 1980s, almost all populations were extinct. Only the remnant population from the Gironde–Garonne–Dordogne basin was still present, with a marine distribution area extending from the Bay of Biscay to the North Sea.

2.5.3.7 Threats

The extinction of sturgeon populations during the last two centuries was caused mainly by overfishing, pollution, river regulation, and hydro construction (Rochard *et al.*, 1990; Debus, 1995). Dams and weirs in rivers can hamper the spawning migration (Fernandez-Pasquier, 2000), whereas degraded water quality could affect early life survival or development (Delage, 2015). Climate change can alter the overall suitability of

formerly frequented habitats, raising questions about where to reintroduce the species (Lassalle et al., 2010). Bycatch at sea can seriously affect survival rates (Rochard et al., 1997). Finally, the introduction of alien sturgeon species could lead to disease transmission and competition (Arndt et al., 2000) as well as confusion in the distinction between protected and alien species when captured incidentally.

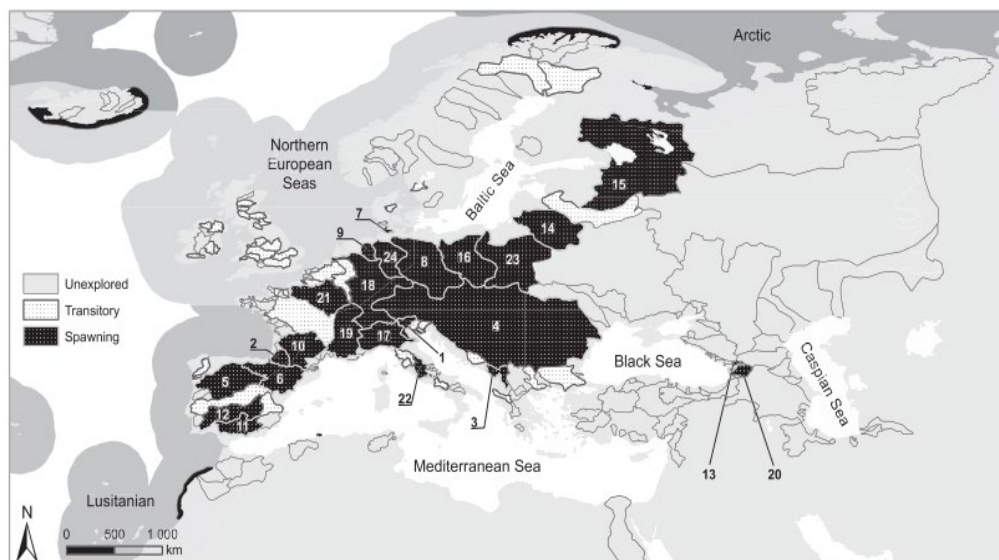


Figure 2.15. Distribution of *Acipenser sturio* around 1750–1850 (from Lassalle et al., 2010). A distinction between the three functional groups of basins is made: unexplored, transitory (individuals observed in the watersheds or within the estuaries, but no young stage), and spawning basins. Basins 15, 14, 16, and 23 represent habitats utilized by *A. oxyrinchus* according to current information.

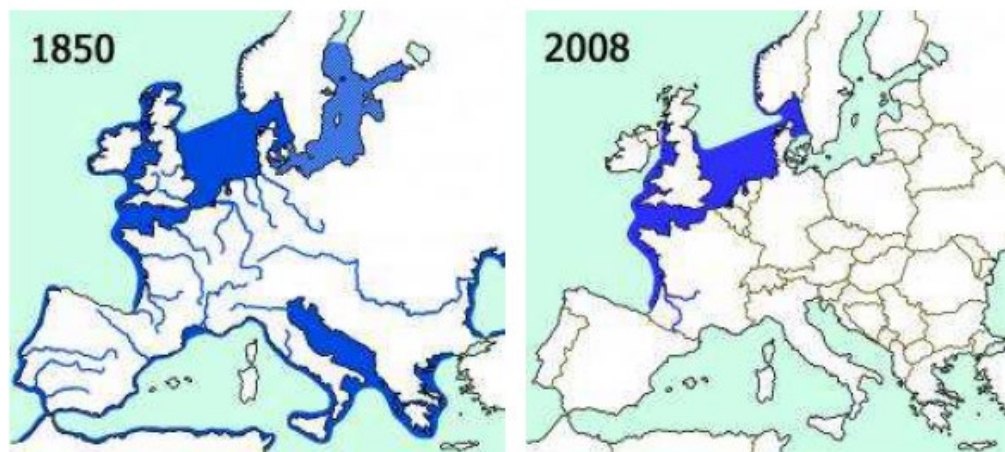


Figure 2.16. Development of *Acipenser sturio* repartition between 1850 and 2008; the lighter shade of blue for 1850 indicates transitory habitat in the Baltic Sea (from MEDDTL, 2011).

2.5.3.8 Conclusion – future

To ensure the viability of the sustained population, long-term and coordinated recovery actions need to be implemented. These have already started thanks to national action plans (Rosenthal et al., 2007; Gessner et al., 2010; MEDDTL, 2011). However, because of the lifespan of the species, long-term recovery can only happen long-term with close international cooperation.

The establishment of *ex situ* stocks was started in France in the early 1990s utilizing wild fish and offspring from a first successful controlled reproduction in 1995. Between 2007 and 2015, releases of early life phases (feeding fry to yearlings) derived from this *ex situ* stock have been carried out under national action plans in France, in the Dordogne and Garonne rivers, and in Germany, in the Elbe River and selected tributaries. A total of 1 650 000 fish have been released in the Gironde system and 20 000 fish in the Elbe River. A backup population from offspring of the controlled reproductions in France has been established under *ex situ* conditions in Germany. Monitoring is carried out on an annual basis in the Gironde River in order to determine the success of the releases and the development of the population. No natural reproduction has yet been confirmed.

Releases and monitoring, as well as information in riverine and coastal fishery, have resulted in capture reports from northern Spain, France, UK, and the North Sea range states. In general, there is a high compliance in fishery, based on the reported catches. Mortalities mainly occur in gillnets and bottom-trawl fishery.

When the fish of the 2007 and subsequent year classes enter maturity, the releases of early life phases will be commenced. Further restoration activities are planned for the Rhine River, Ebro, and Po River. These attempts are restricted by the lack of offspring.

3 Catadromous species

3.1 European eel

3.1.1 Taxonomy

Class:	Actinopterygii
Order:	Anguilliformes
Family:	Anguillidae
Scientific name:	<i>Anguilla anguilla</i>

Subspecies, variations, synonyms

None

Taxonomic notes

None

Common names

Adapted from Froese and Pauly, 2005 - EN: European eel, common eel, river eel, weed eel; FR: Angèle, Anguille d'Europe, Anguille européenne; Anguille jaune, civelle, Lep-tocéphale; ES: Anguila, Anguila europea, Anguilla; PT: Enguia, Enguia europeia; FI: ankerias; SV: ål.

Primary sources of information

Primary sources of information: ICES Stock Annex for European eel (ICES, 2016a); ICES WGEEL report 2015 (ICES, 2015e); IUCN Red List 2014 (Jacoby and Gollock, 2014); and ICES Stock Advice for 2016 (ICES, 2015f).

General characteristics

Length at maturity:	35–120 cm
Weight:	0.2–12 kg
Maximum age:	ca. 60 years
Generation length:	average 15 years

European eel (*Anguilla anguilla*) is a long-lived, semelparous (i.e. spawns only once), and widely dispersed stock. The European eel spawns in the Sargasso Sea. The leaf-shaped larvae, known as leptocephali, drift with the ocean currents from the Sargasso Sea to the continental shelf of Europe and North Africa. There they enter continental waters and metamorphose into glass eels (McCleave *et al.*, 1987; Tesch and Wegner, 1990). This growth stage, known as yellow eel, may take place in marine, brackish (transitional), or freshwaters (Daverat and Tomas, 2006). The yellow eel stage lasts from 2 to 50+ years, with this period being typically shorter in warmer waters and longer in colder waters. Subsequently, they metamorphose into the silver eel stage (Bevacqua *et al.*, 2006). Silver eels then migrate to the Sargasso Sea where they spawn, and are presumed to die after spawning.

The European eel is panmictic, meaning that there is a single, randomly mating population extending throughout its natural range from northern Norway to North Africa, and throughout the Baltic and Mediterranean seas (Als *et al.*, 2011; Pujolar *et al.*, 2014). The consequence of this panmixia is that although any improvements in silver eel numbers would be expected to benefit the spawning stock, management and conservation

efforts in any one area will contribute to the overall recruitment rather than having direct benefits to local waters. This highlights the importance of coordinating actions across the range of the species in order to address current concerns about low abundance.

Estimates at the juvenile glass eel stage indicate that recruitment across Europe fell in the 1980s to about 10% of former levels (a reference period of 1960–1979 is used). A further decline to 1–5% of these levels has occurred in most years since 2000 (ICES, 2015e). As a result of this decline, ICES currently advises that the stock is outside safe biological limits, and that all anthropogenic impacts on eel production should be reduced to as close to zero as possible (ICES, 2015e). The European eel is also listed as CR on the IUCN's Red List (Jacoby and Gollock, 2014) and on Appendix II of CITES. As a consequence, eel trade into or out of the European Economic Zone is effectively banned.

Fisheries operate throughout most of the continental range of the eel, and across a diversity of age classes, from glass eel to silver eel. Other anthropogenic impacts may include barriers to migrations, impacts from passage through pumps and turbines, habitat loss, pollution, and other impacts on habitat quality. Environmental factors, natural and man-made, may also impact eel production.

3.1.2 Life cycle and migrations

European eels have a complex life history, being a long-lived, semelparous, and widely dispersed stock. The shared single stock is panmictic (Palm *et al.*, 2009), and data indicate that the spawning area is in the southwestern part of the Sargasso Sea (McCleave *et al.*, 1987; Tesch and Wegner, 1990). The newly hatched leptocephalus larvae drift with the ocean currents to the continental shelf of Europe and North Africa where they metamorphose into glass eels and enter continental waters. The growth stage, known as yellow eel, may take place in marine, brackish (transitional), or freshwaters. This stage typically lasts 2–25 years (but can exceed 50 years), prior to metamorphosis to the silver eel stage and maturation. Age-at-maturity varies according to temperature, latitude and longitude, ecosystem characteristics, and density-dependent processes. The European eel life cycle is shorter for populations in the southern part of their range compared to the north. Silver eels migrate to the Sargasso Sea where they spawn and die after spawning. This act has not yet witnessed in the wild. It is impossible to determine the number of age groups that contribute successfully to the spawning effort each year, although it seems likely that the number is considerable.

3.1.3 Stock structure and population dynamics

The European eel is panmictic, so there is no lasting genetic structure (Als *et al.*, 2011). However, morphology and life-history patterns vary across the diverse environments in which eel can be found. In warmer, more southerly regions, growth rates are typically faster and age at maturity shorter. Juvenile eels appear to have no gender, but become male or female after a few months or years. Higher densities and higher growth rates tend to result in a greater proportion of juveniles becoming males (Davey and Jellyman, 2005). Males mature at around 35–50 cm total length, whereas females typically mature at lengths > 45 cm (Bevacqua *et al.*, 2006).

The extensive range of age at maturity (silvering), especially among females, gives rise to a broad range of age classes. There are therefore year cohorts within any annual spawning stock.

3.1.4 Genetics

The European eel is panmictic – that is, there is a single randomly mating population (Als *et al.*, 2011). Despite panmixia, Pujolar (2014) showed correlations between allele frequencies at some loci and environmental variables in the European eel, and suggested that the selected polymorphisms are maintained through spatially varying selection. Pavey *et al.* (2015) demonstrated a polygenic basis of freshwater and brackish/saltwater ecotypes in the American eel. Common garden experiments revealed genetic patterns of growth rates related to geographic zones in American eel (Côté *et al.*, 2009, 2014, 2015). In line with these results, Boivin *et al.* (2015) studied how salinity and geographic origin influenced growth and habitat selection in the same species, and showed that differences in growth between glass eels reared in different areas had a genetic basis.

3.1.5 International status with country highlights

3.1.5.1 International

OSPAR Convention:	Listed
EC Habitats Directive:	–
CITES:	Appendix II (listed in 2007, but effective in March 2009 owing to an 18-month delay). In December 2010, the EU-CITES Scientific Review Group (SRG) decided that it was not possible to make a Non-detriment Finding (NDF) for European eel after reviewing the stock. Since that time, trade to or from the EU of specimens of European eel has been prohibited. ICES recently developed advice on potential conditions under which this prohibition might be lifted (ICES, 2015d). Trade from non-EU range states to non-EU countries is still permitted, provided those states have demonstrated NDF.
Bern Convention:	–
Bonn Convention:	Appendix II since 2014, whereby parties to the Convention (covering almost the entire distribution of European eel) call for cooperative conservation actions to be developed among range states
IUCN Criteria:	A2bd+4bd (Jacoby and Gollock, 2014)
Global IUCN Red List Category:	CR
European IUCN Red List Category:	CR
HELCOM Red List:	CR

3.1.5.2 ICES Advice

ICES (2017a) advised that “when the precautionary approach is applied for European eel, all anthropogenic impacts (e.g. recreational and commercial fishing on all stages, hydropower, pumping stations, and pollution) that decrease production and escape-ment of silver eels should be reduced to – or kept as close to – zero as possible”.

The status of eels remains critical. Annual recruitment indices of glass eel to European waters in 2017 remained low, at 1.6% of the 1960–1979 level in the North Sea series, and 8.7% in the “Elsewhere Europe” series (Figure 3.1). The annual recruitment of young yellow eel to European waters was 24% of the 1960–1979 level. The indices are based on data from fishery and scientific surveys, and form the longest and most reliable time-series that can be used as abundance index. The ICES advice on the state of the stock is based on the fact that these indices are still well below the 1960–1979 levels. Total landings and effort data are incomplete. There is a great heterogeneity among the landings time-series owing to inconsistencies in reporting by, and among, countries, as well as incomplete reporting. Changes in management practices have also affected the reporting of non-commercial and recreational fishery.

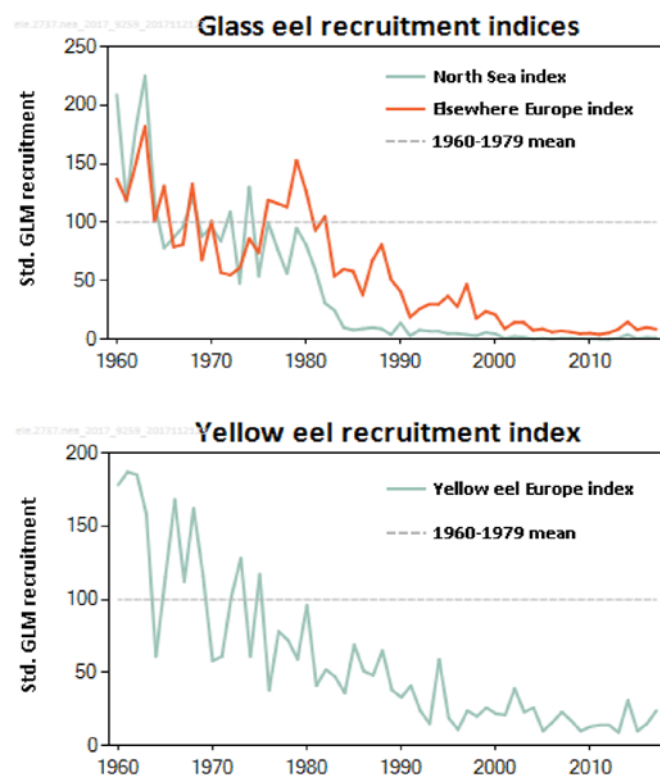


Figure 3.1. Upper panel: recruitment index, geometric mean of estimated (GLM) glass eel recruitment for the continental North Sea and “Elsewhere Europe” series. The GLM (predicting recruitment as a function of area, year, and site) was fitted to 43 time-series, comprising either pure glass eel or a mixture of glass eels and yellow eels, and scaled to the 1960–1979 geometric mean. The “North Sea” series are from Belgium, Denmark, Germany, the Netherlands, Norway, and Sweden. The “Elsewhere” series are from France, Ireland, Italy, Portugal, Spain, and UK. Lower panel: geometric mean of estimated (GLM) yellow eel recruitment trends for Europe. The GLM (predicting recruitment as a function of year and site) was fitted to 14 yellow eel time-series and scaled to the 1960–1979 arithmetic mean (from ICES, 2017a).

3.1.5.3 EU Regulation 1100/2007

The EC Council Regulation 1100/2007 (European Council, 2007) for the recovery of the eel stock required member states to establish eel management plans for implementation in 2009. Under the EC Regulation, member states should: (i) monitor the eel stock, (ii) evaluate current silver eel escapement against escapement that would have occurred if the stock had not been affected by human factors (sometimes called the “pristine”), and (iii) evaluate implemented management actions aimed at reducing eel mortality and increasing silver eel escapement. Under the Regulation, each member state reports to the Commission every third year, starting in 2012.

The Eel Regulation 1100/2007 only applies to EC member states, but the eel distribution extends much farther than this. The international assessment of the whole stock requires data and information from both EU and non-EU countries that produce eels. Some non-EU countries provide data to the WGEEL (ICES Working Group on Eels), and further countries are receiving support to achieve this by the General Fisheries Commission of the Mediterranean (GFCM). GFCM has recently been integrated into WGEEL, with the goal of facilitating knowledge transfer and a full international stock assessment. GFCM is currently undertaking a series of case studies to develop regional multiannual management plans for shared stocks. Coordinated measures, however, must necessarily be simple and adaptable to data-poor situations, given the wide variation in data availability across countries. In the GFCM region, eel is included as one of the priority fishery shared by all countries.

3.1.5.4 Country-by-country status

The Eel Regulation (European Council, 2007) specifies a limit reference point for the escaping silver eel biomass of 40% of what would have existed if no anthropogenic influences had impacted the stock. A lifetime mortality limit $\Sigma A = 0.92$ corresponds to the 40% biomass limit (Dekker, 2010; ICES, 2011a, 2011b). In principal, the international assessment approach consists of the *post hoc* summing up of stock indicators, based on estimates for:

- B_{current} : the amount of silver eel biomass that currently escapes to the sea to spawn, corresponding to the assessment year;
- B_0 : spawner escapement biomass in absence of any anthropogenic impacts;
- B_{best} : spawner escapement biomass, corresponding to recent natural recruitment, that would have survived if there was only natural mortality and no stocking, corresponding to the assessment year;
- ΣA : (i) the sum of anthropogenic mortality rates, i.e. $\Sigma A = \Sigma F$ (the fishing mortality rate, summed over the age groups in the stock.) + ΣH (the anthropogenic mortality rate outside the fishery, summed over the age groups in the stock); or (ii) %SPR, the ratio of actual escapement B_{current} to best achievable spawner escapement B_{best} .

There is currently no complete country by country stock status assessment. However, the following two tables provide the information available in 2014 for eel management units in the European Union (Table 3.1) and for countries bordering the Mediterranean Sea (Table 3.2).

Table 3.1. Summary stock indicators of silver eel biomass (t) and mortality rates for eel management units in 2014 made available to EIFAAC/ICES/GFCM WGEEL in 2015. Biomass and mortality metrics are as explained in text above (from ICES, 2016a). ND = No Data.

Country and EMU code	Biomass (t)				Mortality		
	B ₀	B _{best}	B _{current}	% B ₀	ΣA	ΣF	ΣH
Belgium							
BE_Sche	207	31	23	11	0.30	ND	ND
BE_Meus	32	16	1	3	2.77	ND	ND
Denmark							
DK_Inla	1 110	168	132	12	0.24	ND	ND
France							
FR_Sein	3 925	1 145	172	4	1.89	1.80	0.09
FR_Arto	841	245	13	2	2.93	2.72	0.21
FR_Rhon	642	186	105	16	0.57	0.37	0.20
FR_Rhin	106	31	80	75	1.36	1.26	0.10
FR_Cors	26	8	7	27	0.09	0.06	0.03
FR_Meus	25	7	4	16	0.60	0.30	0.30
FR_Loir	33 857	9 777	155	0	4.15	4.09	0.06
FR_Garo	17 086	4 961	176	1	3.33	3.22	0.11
FR_Adou	4 798	1 411	21	0	4.21	4.10	0.11
FR_Bret	3 974	1 149	46	1	3.21	3.13	0.08
Germany							
DE_Warn	1 499	1 030	996	66	0.10	0.10	0.00
DE_Eide	3 031	835	791	26	0.05	0.01	0.05
DE_Ems	663	156	306	46	0.10	0.09	0.01
DE_Wese	605	74	289	48	0.43	0.23	0.20
DE_Elbe	1 397	54	91	7	1.52	1.25	0.27
DE_Oder	151	7	9	6	1.01	1.01	0.00
DE_Rhei	288	3	146	51	0.92	0.30	0.62
DE_Maas	4	0	0	0	0.51	0.40	0.11
DE_Schl	1 355	1 669	1 657	122	0.05	0.05	0.00
Greece							
GR_NorW	100	31	14	14	ND	ND	ND
GR_WePe	5	21	6	106	ND	ND	ND
GR_EaMT	72	5	2	3	ND	ND	ND
Ireland							
IE_West	192	134	134	70	0.00	0.00	0.00
IE_NorW	136	88	82	60	0.07	0.00	0.07
IE_Shan	201	80	72	36	0.10	0.00	0.10
IE_East	21	15	15	71	0.01	0.00	0.01
IE_SouW	25	14	14	56	0.01	0.00	0.01
IE_SouE	15	10	10	67	0.00	0.00	0.00

Table 3.1 (continued)

Country and EMU code	B₀	B_{best}	B_{current}	% B₀	ΣA	ΣF	ΣH
Italy							
IT_Vene	1 773	411	337	19	0.27	0.13	0.14
IT_Emil	458	123	79	17	0.39	0.30	0.09
IT_Pugl	400	104	78	20	0.24	0.13	0.12
IT_Sard	210	90	21	10	1.18	1.04	0.14
IT_Frio	293	68	52	18	0.39	0.30	0.09
IT_Lazi	71	30	5	7	1.60	1.13	0.47
IT_Tosc	75	27	3	4	2.49	2.46	0.03
IT_Lomb	66	8	4	6	2.91	0.00	2.91
IT_Camp	14	5	4	26	0.18	0.13	0.05
IT_Sici	8	3	2	21	0.31	0.15	0.16
IT_Piem	16	2	0	1	ND	0.02	ND
IT_Tren	7	1	0	1	ND	0.02	ND
IT_Marc	4	1	0	12	0.36	0.10	0.25
IT_Ligu	2	1	0	24	0.28	0.12	0.10
IT_Basi	2	1	0	21	0.21	0.00	0.21
IT_Umbr	4	1	0	0	ND	0.00	ND
IT_Cala	2	0	0	14	0.36	0.20	0.16
IT_Abru	2	0	0	17	0.14	0.00	0.14
IT_Moli	1	0	0	20	0.25	0.00	0.25
IT_Vall	1	0	0	0	ND	0.00	ND
Lithuania							
Lt_Lith	87	18	9	10	0.52	0.42	0.25
Netherlands							
NL_Neth	10 400	1 697	1 057	10	0.47	0.35	0.12
Poland							
PL_Oder	1 611	241	58	4	1.55	1.04	0.51
PL_Vist	1 343	234	33	2	2.31	1.51	0.80
Spain							
ES_Vale	698	407	385	55	0.06	0.06	0.00
ES_Anda	5 563	335	141	3	0.86	0.86	0.00
ES_Basq	245	239	127	52	0.64	0.64	ND
ES_Bale	331	221	221	67	0.00	0.00	ND
ES_Gali	111	52	25	22	0.74	0.72	0.02
ES_Murc	22	47	14	62	1.26	1.26	0.00
ES_Astu	63	44	27	43	0.47	0.47	0.00
ES_Cant	10	37	1	13	3.37	3.37	ND
ES_Nava	5	2	ND		ND	ND	ND
ES_Inne	2 420	0	0	0	ND	ND	ND
ES_Cata	365	ND	ND	ND	ND	ND	ND

Table 3.1 (continued)

Country and EMU code	B ₀	B _{best}	B _{current}	% B ₀	ΣA	ΣF	ΣH
Sweden							
SE_Inla	595	330	91	15	1.29	0.38	0.96
SE_East	12 500	3 770	3 557	28	0.02	0.02	0.00
UK							
GB_NorE	4	ND	ND	ND	0.00	0.00	0.00
GB_Scot	268	454	376	140	0.19	0.00	0.19
GB_Seve	900	384	83	9	1.53	1.20	0.32
GB_SouW	1 328	319	23	2	2.64	2.52	0.12
GB_Neag	500	253	155	31	0.88	1.33	-0.06
GB_NorW	865	205	6	1	1.48	0.59	0.88
GB_Angl	341	172	95	28	0.60	0.09	0.50
GB_Tham	252	162	52	20	1.15	0.04	1.10
GB_Humb	138	138	43	31	1.16	0.05	1.11
GB_SouE	121	76	51	42	0.40	0.04	0.36
GB_Deer	636	53	32	5	0.51	0.04	0.47
GB_Solw	1 474	39	30	2	0.26	0.00	0.26
GB_Wale	430	37	27	6	0.31	0.07	0.24
GB_Nort	61	10	4	7	0.76	0.00	0.76

3.1.6 Distribution summary

Countries in which the European eel is present are (Jacoby and Gollock, 2014): Albania, Algeria, Austria, Belgium, Bosnia and Herzegovina, Bulgaria, Croatia, Cyprus, Czech Republic, Denmark, Egypt, Estonia, Faroe Islands, Finland, France, Georgia, Germany, Gibraltar, Greece, Guernsey, Iceland, Ireland, Isle of Man, Israel, Italy, Jersey, Latvia, Lebanon, Libya, Lithuania, Luxembourg, North Macedonia, Malta, Moldova, Montenegro, Morocco, the Netherlands, Norway, Poland, Portugal, Romania, Russian Federation, Slovakia, Slovenia, Spain, Sweden, Syrian Arab Republic, Tunisia, Turkey, Ukraine, and UK.

The European eel is distributed across the majority of coastal countries in Europe and North Africa. Its southern limit lies in Mauritania (30°N), its northern limit in the Barents Sea (72°N), and it spans all of the Mediterranean basin (ICES, 2014c; Figure 3.2). The spawning area in the Sargasso Sea is thought to be situated on a narrow latitudinal range, 23° and 29.5°N, and a wider longitudinal range, from 48 to 78°W (McCleave *et al.*, 1987; Tesch and Wegner, 1990). At the continental scale, European eels have a wide and scattered distribution, and are found in virtually all types of water bodies from rivers and lakes, to estuaries and coastal waters. Its distribution area is estimated to be ca. 90 000 km² (Moriarty and Dekker, 1997; Dekker, 2009). It is not known what areas contribute to successful spawning or to what degree.

Table 3.2. *Anguilla anguilla* 2014 assessment results for 13 countries in the Mediterranean area: pristine and current wetted area, B_0 , B_{curr} , and B_{best} , stock status as percentage of the pristine escaping biomass (% pristine) and potential escaping biomass (% potential), and lifetime anthropogenic (As with stocking, A without stocking) and fishing mortalities (from ICES, 2016a).

Country	Surface (ha)		Reference points (t)			Stock status (%)		Mortalities (lt-1)		
	Pristine	Current	B_0	B_{best}	$B_{current}$	Pristine	Potential	ΣAs	ΣA	F
Italy	145 852	133 402	4 080.9	891.5	619.2	15.2	69.5	0.34	0.37	0.32
Spain	49 099	22 134	711.3	166.7	81.7	11.5	49.0	0.81	0.83	0.36
France	77 304	69 333	3 596.9	712.5	158.0	4.4	22.2	1.85	1.85	1.80
Algeria	5 723	5 388	156.8	54.0	22.9	14.6	42.5	0.71	0.71	0.70
Albania	58 898	56 458	487.5	221.2	125.6	25.8	56.8	0.55	0.55	0.53
Tunisia	52 073	52 073	1 714.7	337.3	276.2	16.1	81.9	0.18	0.18	0.18
Morocco	12 335	11 600	231.1	138.0	51.5	22.3	37.3	1.26	1.26	1.25
Lybia	3 680	3 680	72.1	22.8	22.7	31.5	99.7	0.00	0.00	0.00
Egypt	379 274	229 044	7 552.0	1 487.0	769.3	10.2	51.7	0.83	0.83	0.71
Turkey	18 205	15 810	322.8	69.6	52.3	16.2	75.1	0.28	0.28	0.20
Greece	57 017	57 017	869.1	187.5	111.3	12.8	59.4	0.56	0.56	0.56
Croatia	1 106	446	7.9	2.5	2.0	25.2	80.0	0.20	0.20	0.00
Montenegro	2 492	2 492	48.8	15.4	15.4	31.4	99.6	0.00	0.00	0.00
Total	696 361	599 761	19 319	4 129	2 199	11.4	53.3	0.68	0.69	0.62

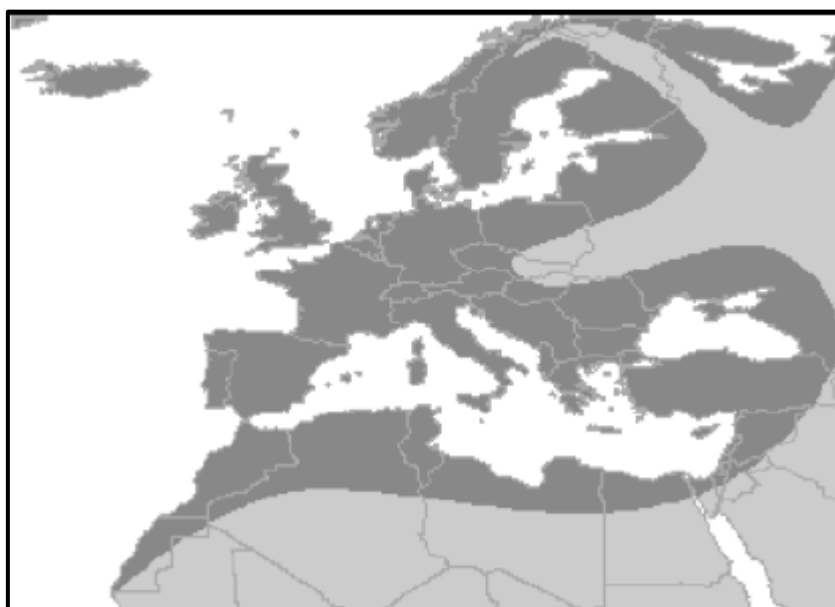


Figure 3.2. The distribution area of European eel (Moriarty and Dekker, 1997).

3.1.7 Country-by-country changes in distribution

No information available.

3.1.8 Threats

The causes of the declining European eel recruitment rates are still not fully understood. There are many hypotheses, but the significance of any single postulated threat, and their potential combined influence, is still poorly understood.

The assessment and management of fishery and non-fishery related mortality factors are carried out by national and regional authorities. Fisheries take place on all continental life stages and throughout the distribution area. Fishing pressure varies from area to area, from almost nil to heavy overexploitation. Illegal, unreported, and unregulated fishing is believed to occur. Non-fishing, anthropogenic mortality factors can be grouped into: (i) hydropower, pumping stations, and other water intakes; (ii) habitat loss or degradation; and (iii) pollution, diseases, and parasites. In addition, anthropogenic action may affect mortality through changes in the abundance of predators, e.g. conservation or culling of predators. Environmental impacts in transitional and freshwaters all contribute to the anthropogenic stresses and mortality of eels and also affect their reproductive success. These include habitat alteration, barriers to eel passage, deterioration in water quality, and presence of non-native diseases and parasites.

Fisheries take place over the entire geographic range of the European eel, and most often are scattered, small-scale, rural enterprises (Dekker, 2004). Eel are traded both locally and internationally. Total landings and effort data are incomplete. There is great heterogeneity among the time-series of landings owing to inconsistencies in reporting by, and between, countries, as well as incomplete reporting. Changes in management practices have also affected the reporting of non-commercial and recreational fishery. Figure 3.3 presents the total landings for all life stages as reported by countries to the WGEEL.

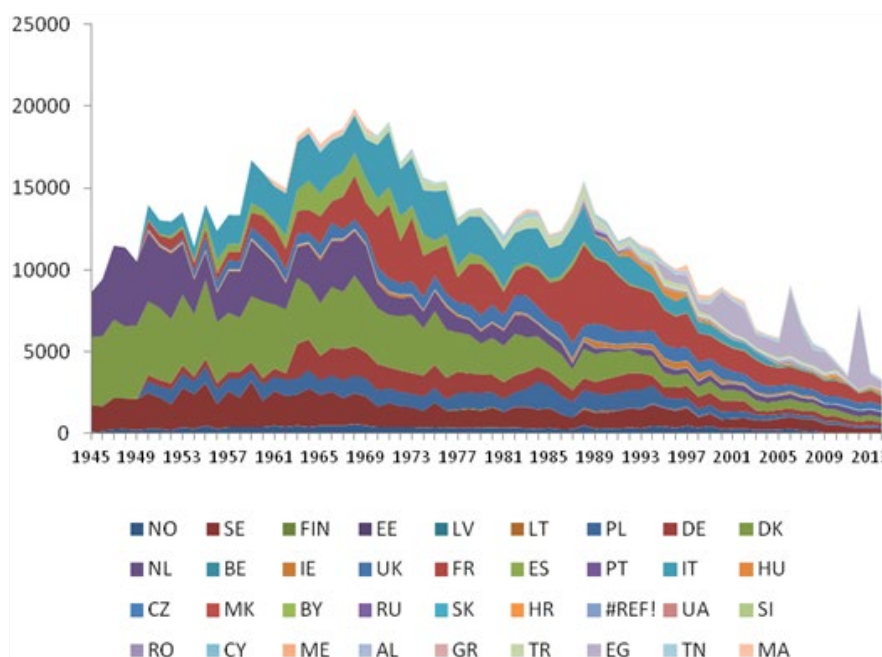


Figure 3.3. *Anguilla anguilla* total landings (all life stages) from 2016 country reports to EI-FAAC/ICES/GFCM WGEEL (not all countries reported); the corrected trend has missing data filled by GLM (ICES, 2013a).

3.1.8.1 Glass eel fishery

The glass eel fishery are mainly concentrated around southwest England, the Bay of Biscay area (Dekker, 2003), and along the Mediterranean coasts of Spain and Italy (ICES, 2012). It is executed in the estuaries and river mouths, fishery and capitalizes on the natural concentration of glass eels in the area (Dekker, 2003). The exploitation of glass eels takes place in winter and early spring when they arrive at the European coast. Fishing is done with both active and passive gears. The active gear includes different hand-held or ship-borne nets, whereas passive gear is composed of traps and fykenets kept fixed in streams (Dekker, 2002). Caught glass eels are used for stocking, aquaculture, or local consumption. The EU Regulation (Article 7.1: European Council, 2007) states that 60% of the eels < 12 cm in length caught annually in EU waters should be reserved for stocking.

3.1.8.2 Yellow eel/silver eel fishery

Yellow and silver eel fishery are located all over the distribution area of the species, from the Mediterranean basin to northern Scandinavia (Dekker, 2003). Some countries, however, have reduced or closed their fishery in response to the EU Regulation. Historically, the biggest landings have been reported from the northern part of the distribution area, with the exception of Spain and Italy in the Mediterranean. Various types of gear are used in the yellow and/or silver eel fishery in both salt and freshwater, including different nets, traps and hooks (Dekker, 2003). The eel fishery located in coastal and rural areas all over Europe are rather small-scaled, making up < 5% of the total European catch (Dekker, 2002). According to Moriarty and Dekker (1997), these fishery employed thousands of people across Europe in the 1990s, but the number is thought to have since declined. In many European countries, yellow and silver eels are not distinguished in the reported catch (ICES, 2014c). Directed fishery for silver eel in coastal waters are specific to the Baltic/Kattegat, where poundnets are used (Dekker, 2003). As

the eel densities are low in the northern areas (25 eels km⁻² of land surface), the fishery is concentrated on the emigration period in late summer and autumn, when most of the silver eel is exploited. In contrast, yellow eel fishery are established where eel densities per km² of land surface are much higher (Dekker, 2003) and fishing seasons extend for more months. Caught yellow and silver eel are mainly sold for consumption either locally or after export to neighbouring countries, mostly within the EU (ICES, 2016a).

3.1.8.3 Recreational fishery

In many EU countries, recreational fishery contributes significantly to the total catch. The gear consists of rod-and-line, longlines, and nets or traps. Usually a licence or permit is required to fish recreationally. However, there are countries where access to the fishery is free or based on private ownership (Dekker, 2005). Data on recreational fishery are collected, but inconsistencies in reporting make assessments unreliable (ICES, 2014c). Overall, the impact of recreational fishery on the eel stock is thought to be of a similar order of magnitude to those of the commercial fishery (ICES, 2017b).

3.1.8.4 Other anthropogenic mortality

In addition to fishery, other forms of anthropogenic mortality exert considerable pressure on the eel stock. These mortalities can be quantified and applied in the reporting of silver eel production, escapement, and mortality, under the requirements of the EU Regulation (European Council, 2007). Obstacles to migration in river systems are one of several factors that cause considerable mortality, and are likely to have contributed to the dramatic decline in the eel population. All continental life-history stages of eel can be impacted by different types of barriers and obstacles. Juvenile eels may be obstructed in their upstream migrations, increasing density-dependence in downstream areas. Silver eels, and large yellow eels in some locations, can be (i) delayed in their downstream migration as a result of river discharge regulation, often leading to changed behaviour and increased predation, and are (ii) likely to experience significant mortality rates associated with passage downstream through power-generation facilities. Pumping stations associated with water-level control and cooling-water intakes are also often a cause of yellow and silver eel mortality. Fish passes are used as an engineered mitigation measure for reducing such impacts, although many studies show that fish passes are not available, not effective, or not working at all.

3.1.8.4.1 Hydropower installations

Hydropower has been recognized as one of several factors contributing to the decline in the eel population (ICES, 2002). Eels tend to have considerably greater mortality rates from downstream passage at hydropower stations than other fish species (Hadderingh and Bakker, 1998). Mortality and injury as a consequence of hydropower stations can occur at inadequate deflection screens, in turbines, and in the tail races. The rate of injury depends on the position of the turbine in the river bed (eels migrate in the main current), the working regime (switching off the turbine during the main migration period reduces the damages), the efficacy of the protection screen, the turbine type, the water flow rate, the head height, characteristics of the turbine, and the presence and location of spillways. Gomes and Larinier (2008) developed mortality predictive equations based on body length of eels, turbine diameter, nominal discharge, and blade velocity for Kaplan turbines. According to this model based on 71 field studies, damage rate increases with fish length, and is generally higher on small turbines with high rotation speeds than on slow, large-diameter turbines. Damage is also lower when

the turbines are fully open (operating at maximum capacity) compared to a reduced opening (Gomes and Larinier, 2008).

Mortality rates when passing a hydropower station also depend on (i) the proportion of eel moving into the power station intake, (ii) the mortality rate of those moving into the power station (turbine mortality, impingement on the trash rack, etc.), and (iii) the mortality rate of those using alternative routes (bypass channels, old river bed, etc.). Mortality estimates of downstream migrating eels from hydropower are given in Table 3.3. The table summarizes field studies from several eel species (*A. anguilla*, *A. rostrata*, *A. dieffenbachia*, and *A. australis*). It should be noted that in many rivers, there are multiple hydropower installations; and, consequently, the cumulative mortality rates are considerable.

Table 3.3. Eel Mortality estimates at hydropower generating plants according to type of turbine and presence of a mitigation systems (bypass, fish-friendly turbine). The number of studies used to calculate the average mortality rates is given in brackets (from ICES, 2011b). Note: there is no direct correspondence between the two columns.

Turbine and mitigating system	Turbine mortality (%)	Total mortality (%)
Average (all turbines)	28 (29)	36 (10)
Average francis	32 (7)	52 (3)
Average kaplan	38 (9)	28 (6)
Average other turbines (mix, propeller, unknown)	21 (11)	40 (1)
Average no bypass or unknown	32 (24)	44 (6)
Average with bypass	9 (5)	26 (4)

3.1.8.4.2 Pumping stations

Pumping stations can negatively influence fish and fish migration, as illustrated in Figure 3.4. First, pumping stations can cause damage, and direct or delayed mortality to fish passing through a pump. Second, a pumping station functions as a potential barrier for the migration of diadromous fish like eel, during both upstream and downstream migration. The resulting congregation of fish often means that they are more susceptible to predation. Damaged and confused fish will be easier prey for piscivorous fish or birds. The risk of being captured by commercial or recreational fishers is higher in the vicinity of pumping stations, where migratory fish aggregate while searching for an opportunity to pass. Various factors, such as pump and propeller type, head of water, capacity, and timing of operation are all known to influence the level of impact on eel (ICES, 2011b). Some impact estimates are summarized in Table 3.4. Buysse *et al.* (2014) demonstrated that propeller pump and Archimedes screw pumps cause eel mortality in lowland canal situations. Buysse *et al.* (2015) assessed maximum mortality rates over a 12-month period ranging from $19 \pm 4\%$ for the large de Wit Archimedes screw pump, to $14 \pm 8\%$ for the small de Wit Archimedes screw pump, based on the condition of the fish and injuries sustained (2012–2013).

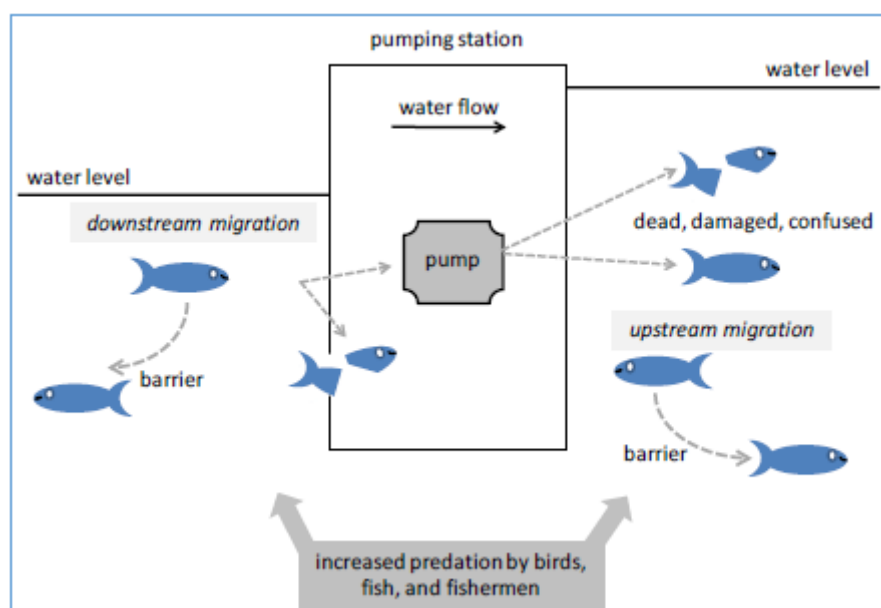


Figure 3.4. Impacts of pumping stations on fish and fish migration (redrawn from STOWA, 2010).

Table 3.4. Mortality estimates of eel passing through pumping stations of various types. The number of studies used to calculate the average mortality rates is given in brackets (summarized from ICES, 2011b). Some additional mortality as a result of undetected internal injury may have occurred in a few studies. Note: there is no direct correspondence between the two columns.

Type of pumping station	Damaged (%)	Mortality (%)
Average (all pumps)	30 (18)	26 (27)
Average water wheel	0 (1)	0 (1)
Average Archimedes	12 (4)	5 (7)
Average centrifugal	1 (3)	13 (4)
Average turbine-Archimedes	0 (1)	0 (1)
Average propeller-centrifugal	-	11 (2)
Average propeller	60 (8)	60 (9)
Average propeller (closed)	-	35 (2)
Average hidostal pump	<3 (1)	0 (1)

3.1.8.4.3 Water intakes

Intakes used for water supply represent another anthropogenic threat to aquatic ecosystems and fish stocks. When water is abstracted from surface water bodies, there is a risk that fish and other organisms will be drawn in. This may prevent fish from migrating effectively and lead to fish death or injury at screens, turbines, and pump mechanisms (Environment Agency UK, 2011). Eels can get caught up in intake flows and screens at any stage of their life. However, they are most at risk during their upstream and downstream migrations within freshwater (Environment Agency UK, 2011). The degree of risk or damage is highly site-specific and depends largely on the actual conditions at each location (e.g. type of power plant or technical facility in general, capacity of water intake, configuration and design of mitigation measures including screens and behavioural deterrent systems, and biological characteristics of the potentially impacted species). It should also be noted that outfall sources can also divert and delay eel migrations leading to additional mortality.

- Intakes: Adult silver eels are particularly vulnerable when they actively follow currents downstream (“positive rheotaxis”). Glass eel and elvers are also at risk when they have to pass areas with intakes, which sometimes have enormous capacities for water intake.
- Outfalls: Juveniles (glass eels, elvers, or smaller yellow eels) are more at risk during active migration upstream (“negative rheotaxis”).

3.1.8.4.4 Pollution and parasites

The accumulation of lipophilic chemical pollutants by maturing eels could have potentially toxic effects on migrating adults. These chemicals are stored by the fish and released when fat stores are broken down during migration. This could lead to metabolic disruption, which could limit the capacity of the silver eels to complete their spawning migrations (Robinet and Feunteun, 2002; Palstra *et al.*, 2006). Further, there is concern that even if the spawning migration is completed, lipid stores containing xenobiotics may result in disrupted gonadogenesis and/or low quality gametes (Robinet and Feunteun, 2002).

The parasite nematode (*Anguillicola crassus*), introduced when the Japanese eel (*A. japonica*) was imported to Europe for culture in the early 1980s, is also thought to impact the ability of the European eel to reach its spawning grounds. This parasite has a negative influence on fitness traits associated with the silvering stage of maturation, and it causes swimbladder damage, which impairs swimming performance (Palstra *et al.*, 2007) and the ability to cope with high pressure during its reproductive migration (Vettier *et al.*, 2003; Sjöberg *et al.*, 2009).

3.1.8.4.5 Climate change

Climate change has been proposed to play a role in the fluctuations of in *A. Anguilla* abundance, particularly larval transport and glass eel recruitment. Climate change impact occurs through its effects on the suspected breeding grounds (Sargasso Sea), and through changing oceanic conditions that can influence the recruitment of glass eels to nearshore and freshwater environments. An important consideration in this discussion is the time-scale over which changes are thought to occur as a result of oceanic conditions.

The North Atlantic Oscillation (NAO) has been studied as a recruitment driver for both the European and American eel, with published literature arguing for and against this hypothesis. Durif *et al.* (2011) indicated that periods of high NAO appear to negatively correlate with recruitment to freshwater habitats, as a consequence of the larvae being driven into colder water, which considerably slows down the process of metamorphosis into glass eels. Further, changing ocean climate may be responsible for fluctuations in ecosystem productivity and, thus, food availability for leptocephali (Miller, 2009).

3.1.9 Conclusions – future

It is very difficult to predict the future state of the European eel, given the uncertainties over stock dynamics, the impacts of diverse anthropogenic factors, and the effects of implemented or proposed management measures. However, international recruitment indices have yet to show a significant and sustained increase.

Given that total landings and effort data are incomplete, ICES lacks the information needed to provide a reliable estimate of total catches of eel. Furthermore, the understanding of the stock dynamic relationship is not sufficient to determine/estimate the impact of any catch above zero (at glass, yellow, or silver eel stage) on the reproductive

capacity of the stock. The focus of the WGEEL in the coming years will be on the following key areas:

- Source the appropriate assessment data across the range of the European eel, by working with the EU, EIFAAC, ICES, and GFCM members.
- Further develop eel-specific stock assessment methods.
- Contribute to the development of a standardized and unified assessment process across the entire distribution of the European eel, working with EU, EIFAAC, ICES, and GFCM members.
- Focus management advice on the pragmatic use of mortality indicators (immediate impact) as short-term goals, leaving biomass indicators (long-term impact) for the longer term goals.

3.2 European flounder

3.2.1 Taxonomy

Class:	Actinopterygii
Order:	Pleuronectiformes
Family:	Pleuronectidae
Scientific name:	<i>Platichthys flesus</i> (Linnaeus, 1758)

Subspecies, variations, synonyms

None

Taxonomic notes

None

Common names

From Fishbase, 2017 – EN: Butt, European flounder, fluke or mud flounder, river flounder, white fluke; DA: Flynder, Skrubbe; NL: Bot; FI: Kampela; FR: Flet, Flet d'Europe; PL: Stornia; PT: Patruça, Petruca, Solha, Solha-das-pedras; ES: Platija, Platija Europea, Plana, Platixa, Solla; SV: Flundra, Skrubba, Skrubblundra, Skrubbskädda

General characteristics

Maximum total length:	60 cm
Maximum reported weight:	2.9 kg
Maximum reported age:	15 years

The European flounder is a demersal species present both in coastal and estuarine ecosystems along the Northeast Atlantic Ocean, from Norway to Portugal (Nielsen, 1986). It is also present in the adjacent seas: Baltic, North, White, Mediterranean, and Black seas (Nielsen, 1986; Figure 3.5, Table 3.5). The species was introduced in North America through ballast water. However, the last record of the species there dates back to 2000 (Erling Holm, pers. comm.).

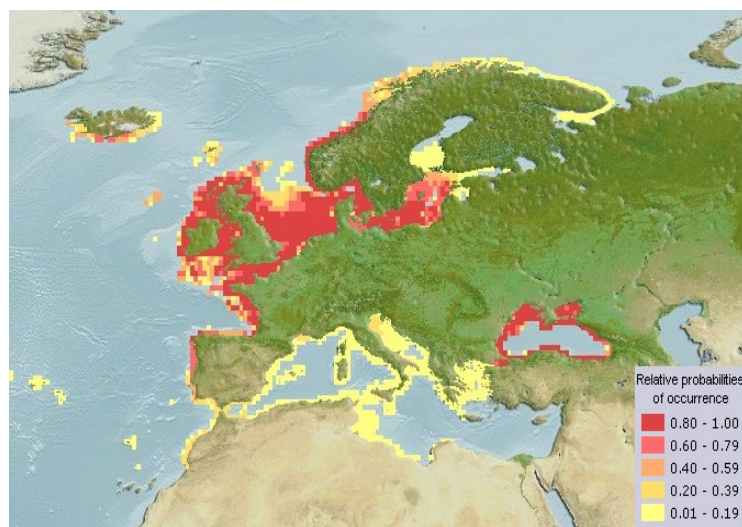


Figure 3.5. Distribution range of the European flounder (*Platichthys flesus* Linnaeus, 1758). Image modified from <http://www.fishbase.org> to include the species presence in the Dvina Bay (White Sea) (Ovsepyan *et al.*, 2014).

The European flounder was previously considered a strict catadromous species. However, there is some controversy on whether to classify flounder as catadromic, semi-catadromic, estuarine migrant, marine-estuarine opportunistic, or even as an estuarine resident species (Elliott and Dewailly, 1995; Elliott *et al.*, 2007). Recent work revealed that flounder is not an estuarine resident species (Daverat *et al.*, 2011a; Morais *et al.*, 2011), despite using estuarine habitats for extended periods of time (Jagger, 1998; Marchand *et al.*, 2003).

3.2.2 Life cycle and migrations

Reproduction occurs in marine waters between January and July. The timing of reproduction varies along its distribution range, starting earlier at lower latitudes (Martinho *et al.*, 2013). It has been suggested that the European flounder might also spawn in estuarine habitats, based on either anecdotal evidence from fishers (Elbe River, Germany (Bos, 1999); Minho estuary and Ria de Aveiro, Portugal) or otolith chemistry studies (Daverat *et al.*, 2012).

Generally, larvae ingress into estuaries before settlement (Jager, 1998; Bos, 1999; Amorim *et al.*, 2016), although some exceptions have been observed along its distribution range (Daverat *et al.*, 2012; Primo *et al.*, 2013). In fact, larvae can use coastal, brackish, and freshwater habitats during larval development and can spend distinct periods of time in each (Daverat *et al.*, 2012). Within estuaries, larvae most likely migrate to nursery grounds using selective tidal stream transport (Bos, 1999). Settlement occurs after metamorphosis, which is a period when the pelagic larvae undergo anatomical modelling and physiological transformations as an adaptation to the benthic habitat (e.g. the head is remodelled asymmetrically and one eye is repositioned to the opposite side; Schreiber, 2006). Juveniles develop in estuarine nurseries, where they can remain until reaching sexual maturity at the age of 2–3 years. Subsequently they migrate to coastal areas (Summers, 1979; Bos, 1999; Dreves *et al.*, 1999).

3.2.3 Genetics

There is genetic structuring among European flounder populations across its distribution (Borsa *et al.*, 1997; Hemmer-Hansen *et al.*, 2007). The subspecies *Platichthys flesus luscus* and *Platichthys flesus italicus* were described for the Baltic Sea and the Adriatic Sea, respectively (Norman, 1934 in Borsa *et al.*, 1997). Later, Borsa *et al.* (1997) identified that the Aegean Sea–Marmara Sea–Black Sea region harbours a single population of *P. flesus luscus*, which is genetically different from the one in the Adriatic Sea. In addition, the populations of *Platichthys flesus flesus* from the Atlantic and the western Mediterranean show a large enough genetic variability to recognize them as separate subspecies (Borsa *et al.*, 1997). Further north, in the White Sea, a further species can be found *Platichthys flesus bogdanovi* (Ovsepyan *et al.*, 2014).

The European flounder shows different life-history strategies, some of which may be linked to genetic variability among populations. The most well-studied case are the Baltic Sea populations, which show two distinct reproductive behaviours: offshore spawning with pelagic eggs, and coastal spawning with demersal eggs (Nissling *et al.*, 2002). Pelagic spawning occurs when salinity is lower than 11 ppt in deep offshore southern and central areas, whereas demersal spawning occurs in shallow coastal areas when salinity reaches 6 ppt (Nissling *et al.*, 2002). These two groups represent a pair of closely related species arising from a speciation event that occurred ca. 8500 years ago (Momigliano *et al.*, 2017).

3.2.4 Stock structure and population dynamics

The European flounder was historically distributed from the Mediterranean Sea to the Baltic Sea, but its recent presence in Iceland suggests a northward expansion (O'Farrel, 2012). Although this species is common along its distribution range, several populations may be decreasing, especially in the extreme limits of its distribution. Landings have decreased in the southern distribution limit in the Northeast Atlantic since the early 1970s, but a northward retraction in its distribution has also been reported (Teixeira *et al.*, 2014). The species has decreased in abundance or disappeared from some southern areas, namely from the Tagus and Sado estuaries (Portugal). In the northern Baltic Sea, i.e. northern distribution limit, a decline has been observed in landings, 46–97% from the 1990s to the 2000s, and flounder size and condition (Jokinen *et al.*, 2015).

3.2.5 International status

The European flounder has been assessed as LC (Least concern) by the IUCN in the IUCN red list of threatened species (Munroe, 2010).

This species is common in most areas of its broad geographic range, and is not threatened, despite some local anthropogenic stressors (e.g. overfishing and pollution; Munroe, 2010).

3.2.6 Threats

The European flounder is harvested by commercial and recreational fishers. The most important fisheries exist in Baltic and Danish waters (Munroe, 2010). Flounder populations have been decreasing in the southern limits of the species, which may compromise the sustainability of local fisheries despite the absence of known instances of over-exploitation.

The species has been impacted by chemical pollution in sediments. Specifically, xeno-oestrogens, originated from domestic, industrial, or agricultural waste, may cause intersex (Munroe, 2010). The synergy of contaminants with the hypoxic conditions observed in many estuaries and coastal areas may cause the future extirpation of local populations.

Climate change, and namely global warming, may be causing a reduction in the historical distribution in the southern range of the species, while expanding the distribution northward. Flounder has been reported recently in Iceland (O' Farrel, 2012).

Flounders used to be reported far inland before river barriers prevented them from colonizing freshwater habitats. These habitats are currently lost for flounders because this species does not have the swimming aptitude to use the conventional fish ladders designed for salmonids.

3.2.7 Distribution

Native: Albania, Algeria, Belgium, Bosnia and Herzegovina, Bulgaria, Croatia, Denmark, Egypt, Estonia, Finland, France, Germany, Gibraltar, Greece, Ireland, Israel, Italy, Latvia, Lebanon, Libya, Lithuania, Malta, Morocco, the Netherlands, Norway, Poland, Portugal, Romania, the Russian Federation, Slovenia, Spain, Sweden, Syrian Arab Republic, Tunisia, Turkey, Ukraine, and UK.

Regionally extinct: Czech Republic.

Introduced: Canada, USA, and Iran. Not established in North America.

A detailed list of the country-by-country distribution of flounder can be seen in Table 3.5.

Table 3.5. Distribution of the European flounder (*Platichthys flesus* Linnaeus 1758) by country.

Country	Distribution
Albania	Data not available
Algeria	Data not available
Belgium	North Sea - de Clerck <i>et al.</i> (1984); Misra <i>et al.</i> (1989, 1990); Roose <i>et al.</i> (1998); Baeyens <i>et al.</i> (2003) Scheldt estuary - Álvarez-Muñoz <i>et al.</i> (2015); Hampel <i>et al.</i> (2005); Maes <i>et al.</i> 1997, 2003); van Ael <i>et al.</i> (2012)
Bosnia and Herzegovina	Data not available
Bulgaria	Data not available
Croatia	Data not available
Denmark	Baltic Sea - Skall <i>et al.</i> (2000, 2005); Madsen <i>et al.</i> (2006); Satomi <i>et al.</i> (2006); Vorkamp <i>et al.</i> (2011) Bornholm Basin: Kijewska <i>et al.</i> (2009); Baršienė <i>et al.</i> (2012, 2014) Faroe Islands - Hemmer-Hansen <i>et al.</i> (2007) Great Belt - Jørgensen and Pedersen (1994); Vorkamp <i>et al.</i> (2011) Bay of Aarhus: Tarpgaard <i>et al.</i> (2005); Madsen <i>et al.</i> (2013) Gabet, Kerteminde, Odense Fjord, Seden, Vejle: Madsen <i>et al.</i> (2013) Kattegat - Skall <i>et al.</i> (2000, 2005); Strand and Jacobsen (2005) Ålborg, Sebbesund: Madsen <i>et al.</i> (2013) Mariager Fjord: Andersen <i>et al.</i> (2005) Little Belt Falshoeft Channel, Rise Channel, Vejsnaes: Petereit <i>et al.</i> (2014) North Sea - Borsa <i>et al.</i> (1997); Mortensen <i>et al.</i> (1999); Skall <i>et al.</i> (2000, 2005) Limfjord: Kiørboe <i>et al.</i> (1983); Andersen <i>et al.</i> (2005); Riisgård <i>et al.</i> (2012); Tomczak <i>et al.</i> (2013) Ringkøbing: Madsen <i>et al.</i> (2013) Thyborøn: Kijewska <i>et al.</i> (2009) Wadden Sea: Vorkamp <i>et al.</i> (2011) Øresund - Køie (2001); Nissling <i>et al.</i> (2002); Køie <i>et al.</i> (2004); Vorkamp <i>et al.</i> (2011)
Egypt	Data not available
Estonia	Baltic Sea - Pandelova <i>et al.</i> (2008) Matsalu Bay: Vetemaa <i>et al.</i> (2006) Off Kalana, off Kelnase, off Spithami: Køie (1999) Saarnaki Islet: Kreitsberg <i>et al.</i> (2010) Vilsandi National Park: Ott (2001) Gulf of Finland - Baršienė <i>et al.</i> (2012); Järv <i>et al.</i> (2017) Keibu Bay: Kreitsberg <i>et al.</i> (2010) Muuga Bay: Dreves <i>et al.</i> (2007) Tallinn Bay: Bogovski <i>et al.</i> (2002); Dreves <i>et al.</i> (2007) Gulf of Riga - Baršienė <i>et al.</i> (2012)

Table 3.5 (continued)

Finland	Baltic Sea
	Åland Islands (ICES 1983)
	Hinderbengtssviken: Schrandt <i>et al.</i> (2016)
	Baltic Sea (continued)
	Nåtö: Voigt (2001, 2014a, 2014b)
	Gulf of Finland - Aro and Sijblom (1983)
	Emska: Karlsson <i>et al.</i> (2003)
	Furuskar: Sipiä <i>et al.</i> (2001a)
	Hangö: Sipiä <i>et al.</i> (2001a); Karlsson <i>et al.</i> (2003); Malmström <i>et al.</i> (2004)
	Knyllgorna, Segelska: Sipiä <i>et al.</i> (2006)
	Kyrksundet: Voigt (2001)
	Sundholm Bay: Sipiä <i>et al.</i> (2001b, 2002)
	Tvärminne: Boström <i>et al.</i> (2002); Kankaanpää <i>et al.</i> (2005); Sipiä <i>et al.</i> (2006); Borg <i>et al.</i> (2014); Voigt (2014a)
	Gulf of Bothnia - Aro and Sijblom (1983)
	Kokemäenjoki-Kumoläv: Voigt (2014a)
	Uusikaupunki: Sipiä <i>et al.</i> (2001)
France	Britany Peninsula - Borsa <i>et al.</i> (1997)
	Canche Estuary - Selleslagh and Amara (2008, 2015); Calvès <i>et al.</i> (2013)
	Loire Estuary - F. Daverat, IRSTEA, France, pers. obs.)
	Gironde Basin
	Dronne River: Daverat <i>et al.</i> (2011a); Le Pichon <i>et al.</i> (2014)
	Garonne River: Daverat <i>et al.</i> (2011b)
	Isle-Saint-Georges (Saucats): Selleslagh <i>et al.</i> (2016)
	Gironde Estuary: Pasquaud <i>et al.</i> (2010); Daverat <i>et al.</i> (2011a, 2012); Selleslagh <i>et al.</i> (2015)
	Isle River: Le Pichon <i>et al.</i> (2014)
	Lary Stream: Le Pichon <i>et al.</i> (2014)
	Gulf of Lion - Borsa <i>et al.</i> (1997)
	River Bidasoa - Lekuona and Campos (1996, 1997)
	Seine Basin
	Coastal area adjacent to the Seine Estuary: Daverat <i>et al.</i> (2012)
	Seine Estuary: Daverat <i>et al.</i> (2012); Calvès <i>et al.</i> (2013)
	Slack Estuary - Martinho <i>et al.</i> (2013)
	Vilaine Estuary - Calvès <i>et al.</i> (2013)
Germany	Baltic Sea - Baršienė <i>et al.</i> (2012)
	Kiel Bay, Lübeck Bay: Borsa <i>et al.</i> (1997)
	Kiel Fjord: Voigt (2014a)
	Mecklenburg Bight, off Hiddensee: Køie (1999)
	Wismar Bay
	Offentief, Walfisch, Wismar Harbour: Lang <i>et al.</i> (2006)
	Elbe River: Bos (1999)
	Little Belt
	Falshoef Channel, Rise Channel, Vejsnaes: Petereit <i>et al.</i> (2014)
	North Sea
	Helgoland Bay, off Rhine Estuary: Borsa <i>et al.</i> (1997)
	Tiefe Rinne: Broeg (2010)

Table 3.5 (continued)

Gibraltar	Data not available
Greece	Thermaikos Gulf - Borsa <i>et al.</i> (1997)
Iceland	Önundarfjörður Hafnarós, Hestá, Korpa, Vatnið, Vöð: O'Farrel (2012)
Ireland	Southeast coast Dublin Bay (North Bull Island): Koutsogiannopoulou and Wilson (2007) Southwest coast Dingle Peninsula (Smerwick), Galway Bay (Ballyloughaun): Haynes <i>et al.</i> (2008)
Israel	Data not available
Italy	Adriatic Sea Po Delta: Viganò <i>et al.</i> (2001) Marano Lagoon: Borsa <i>et al.</i> (1997) Venice Lagoon: Franco <i>et al.</i> (2010); Zucchetto <i>et al.</i> (2010); Pravoni <i>et al.</i> (2013)
Latvia	Baltic Sea Liepāja: Kõie (1999); Barda <i>et al.</i> (2015) Pavilosta: Barda <i>et al.</i> (2015) Ventspils: Kõie (1999); Barda <i>et al.</i> (2015) Gulf of Riga - Baršienė <i>et al.</i> (2012) Kolka: Barda <i>et al.</i> (2015)
Lebanon	Data not available
Libya	Data not available
Lithuania	Baltic Sea - Baršienė <i>et al.</i> (2012) Būtingė: Baršienė <i>et al.</i> (2006); Lang <i>et al.</i> (2006) Nemirseta: Baršienė <i>et al.</i> (2006); Lang <i>et al.</i> (2006) Off Klaipėda: Kõie (1999); Baršienė <i>et al.</i> (2006) Palanga: Baršienė <i>et al.</i> (2006); Lang <i>et al.</i> (2006)
Malta	Data not available
Morocco	Data not available
Netherlands	Wadden Sea Amsteldiep, Borndiep, Breezanddijk, Den Oever, Harlingen, Koehool, Kornwerderzand, Lauwers, Lauwersoog: Vethaak (1992, 2013) Balgzand: van der Veer <i>et al.</i> (2011) Lake Lauwers (Lauwersoog): Vethaak (1992, 2013) Ems-Dollard Estuary - Stronkhorst (1992); Jager (1998) Bight of Watum, Mouth of Dollard, Nieuwe Statenzijl: Vethaak (1992, 2013) North Sea Amsterdam and Rotterdam harbours and adjacent coastal areas, off Noordwijk: de Boer <i>et al.</i> (2001) Lake Grevelingen - Beyst <i>et al.</i> (1999) Oosterschelde - Beyst <i>et al.</i> (1999) Scheldt Estuary - Stronkhorst (1992); Hampel <i>et al.</i> (2004)
Norway	Bergen - Solbakken and Palmork (1981); Goksøyr <i>et al.</i> (1996) Eidangerfjord - Ruus <i>et al.</i> (2006); Hylland <i>et al.</i> (2006) Frierfjord - Ruus <i>et al.</i> (2006); Hylland <i>et al.</i> (2006)

Table 3.5 (continued)

Norway (cont.)	North Sea
	Karmsund: Baršienė <i>et al.</i> (2004)
	Skagerrak
	Hvaler Archipelago: Goksøyr <i>et al.</i> (1991); Marthinsen <i>et al.</i> (1991); Staveland <i>et al.</i> (1993)
	Glomma Estuary: Marthinsen <i>et al.</i> (1991); Staveland <i>et al.</i> (1993)
	Hubukta, Øra: Hylland <i>et al.</i> (1998)
	Oslofjord: Reiersen and Fugelli (1984); Goksøyr <i>et al.</i> (1996); Husøy <i>et al.</i> (1996)
	Balsfjorden (Tromsø) - Lønning <i>et al.</i> (1988)
	Borgenfjord - He and Mork (2015)
	Hardangerfjord
	Eikhamran, Granvin, Grimo, Herand, Nordheimsund, Strandebar, Utne, Varaldsøy: Julshamn and Grahl-Nielsen (1996)
	Sørfjorden - Beyer <i>et al.</i> (1996)
	Trondfjord - Lile (1998)
Poland	Baltic Sea - Baršienė <i>et al.</i> (2012)
	Off Kolobrzeg: Køie (1999)
	Off Leba: Køie (1999)
	Bay of Gdańsk - Køie (1999); Skwarzec <i>et al.</i> (2001); Kijewska <i>et al.</i> (2009); Baršienė <i>et al.</i> (2012)
	Mechelinki, Sobieszewo: Lang <i>et al.</i> (2006)
	Gdańsk: Mazur-Marzec <i>et al.</i> (2007)
	Sopot: Lang <i>et al.</i> (2006); Mazur-Marzec <i>et al.</i> (2007)
Portugal	Douro Estuary - Vinagre <i>et al.</i> (2008)
	Lima Estuary - Ramos <i>et al.</i> (2006); Mendes <i>et al.</i> (2014); Amorim <i>et al.</i> (2016)
	Minho Estuary - Morais <i>et al.</i> (2011); Souza <i>et al.</i> (2013); Dias <i>et al.</i> (2017)
	Mondego Estuary - Calvès <i>et al.</i> (2013); Martinho <i>et al.</i> (2013); Primo <i>et al.</i> (2013)
	North coast - Sobral (2007); Vasconcelos <i>et al.</i> (2008)
	Ria de Aveiro Lagoon - Sobral (2007)
	Sado Estuary - Borsa <i>et al.</i> (1997); currently absent or extremely rare
	Tagus Estuary - Borsa <i>et al.</i> (1997); currently absent or extremely rare
Romania	Data not available
Russian Federation	White Sea
	Dvina Bay: Ovsepyan <i>et al.</i> (2014)
	Dvina River: Ovsepyan <i>et al.</i> (2014)
Slovenia	Data not available
Spain	Bay of Biscay
	River Bidasoa: Lekuona and Campos (1996, 1997)
	Cantabric Sea
	Burela: Bouza <i>et al.</i> (2002)
	River Masma: Fernández-Parajes and Riesco-Muñoz (2016)
	Galician Rías
	Ría de Arousa: Bouza <i>et al.</i> (2002)
	Ría de Muros e Noia: Álvarez <i>et al.</i> (2002)
	Mediterranean Sea
	Ebro Delta: Borsa <i>et al.</i> (1997); Parera <i>et al.</i> (2013)
	Western Atlantic Coast
	Minho Estuary: Morais <i>et al.</i> (2011); Daverat <i>et al.</i> (2012)

Table 3.5 (continued)

Sweden	<p>Baltic Sea</p> <p>Arkona, Bornholm, and Gotland Basins: Nissling <i>et al.</i> (2002)</p> <p>Himmerfjärden: Elmgren <i>et al.</i> (1983)</p> <p>Island Gotska Sandön: Florin <i>et al.</i> (2013)</p> <p>Karlskrona, Nynäshamn, Öland, Västervik: Fornbacke <i>et al.</i> (2002)</p> <p>Kvädöfjärden: Sipiä <i>et al.</i> (2001a); Lang <i>et al.</i> (2006)</p> <p>Tvären Bay: Johansson-Sjöbeck and Larsson (1978)</p> <p>Kattegat</p> <p>Kungsbackafjorden: Florin and Höglund (2008)</p> <p>Øresund - Køie(2001); Nissling <i>et al.</i> (2002); Køie <i>et al.</i> (2004); Vorkamp <i>et al.</i> (2011)</p> <p>Skagerrak</p> <p>Lysekil, Strömstad: Fornbacke <i>et al.</i> (2002)</p> <p>Gullmarn Fjord: Modin and Pihl (1996)</p> <p>Hvaler Archipelago: Goksøyr <i>et al.</i> (1991)</p>
Syrian Arab Republic	Data not available
Tunisia	Data not available
Turkey	<p>Aegean Sea</p> <p>Homa Lagoon: Acarli <i>et al.</i> (2014)</p> <p>Black Sea</p> <p>Samsun: Tsagarakis <i>et al.</i> (2015)</p> <p>Sarıkum Lagoon Lake: Öztürk and Özer (2010)</p> <p>Trabzon: Çiloğlu (2005); Aydın (2012)</p> <p>Mediterranean Sea</p> <p>Dalyan Lagoon: Aydogdu and Öztürk (2003)</p> <p>Sea of Marmara</p> <p>Ekinli Lagoon: Oguz and Öktener (2007)</p> <p>Gönen River: Fautz (1986)</p>
Ukraine	Data not available
UK	<p>England</p> <p>Liverpool Bay, Lyme Bay, Red Wharf Bay, River Dee, River Humber, River Tees, River Thames, River Wear, Rye Bay: Matthiessen <i>et al.</i> (1998)</p> <p>River Alde: Matthiessen <i>et al.</i> (1998); Kirby <i>et al.</i> (2006)</p> <p>River Mersey: Matthiessen <i>et al.</i> (1998); Kirby <i>et al.</i> (2006)</p> <p>River Otter Kennedy (1996); O'Mahony <i>et al.</i> (2004)</p> <p>River Tamar: (Matthiessen <i>et al.</i> (1998); Calvès <i>et al.</i> (2013)</p> <p>River Tyne: Matthiessen <i>et al.</i> (1998); Kirby <i>et al.</i> (2006)</p> <p>Severn Estuary: Claridge <i>et al.</i> (1986); Potter <i>et al.</i> (2001)</p> <p>Southampton Water: Matthiessen <i>et al.</i> (1998); Grinwis <i>et al.</i> (2000)</p> <p>Off Thames Estuary: Borsa <i>et al.</i> (1997)</p> <p>English Channel - Matthiessen <i>et al.</i> (1998)</p> <p>Isle of Man</p> <p>Port Erin: Kirby <i>et al.</i> (2006)</p> <p>Northern Ireland</p> <p>River Bann: Wirjoatmodjo and Pitcher (1984)</p> <p>Scotland</p> <p>Aberdeen: Borsa <i>et al.</i> (1997)</p> <p>Firth of Clyde: Thurstan and Roberts (2010)</p>

Table 3.5 (continued)

UK (cont.)	Scotland (cont.)
	River Clyde: Matthiessen <i>et al.</i> (1998)
	West coast: Craik and Harvey (1986)
	Ythan Estuary: Summers (1979)
	Southern North Sea - Matthiessen <i>et al.</i> (1998)
	Wales
	Cardigan Bay, Carmarthen Bay, River Dee: Matthiessen <i>et al.</i> (1998)
	Severn Estuary: Claridge <i>et al.</i> (1986); Potter <i>et al.</i> (2001)

3.2.8 Fisheries and management

Between 1950 and 2015, the total reported landings of the European flounder varied between 7500×10^3 kg (1954) and $24\,467 \times 10^3$ kg (2005). The increase in total reported landings in the period 2004–2015 are due mainly to the reported Polish landings, which were absent from the FAO database before 2004 (Figure 3.6 and 3.7).

The majority of the reported landings of European flounder occur in countries fishing in the Baltic Sea. Poland (50.5%), Denmark (10.9%), and The Netherlands (10.8%) were the top three countries regarding the reported landings for the period 2006–2015 (Figure 3.8).

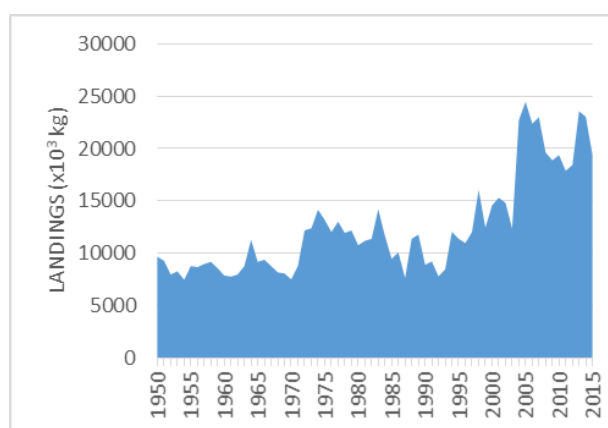


Figure 3.6. Total reported landings ($\times 10^3$ kg) of European flounder (*Platichthys flesus* Linnaeus, 1758) between 1950 and 2015. Data retrieved from FAO (2017).

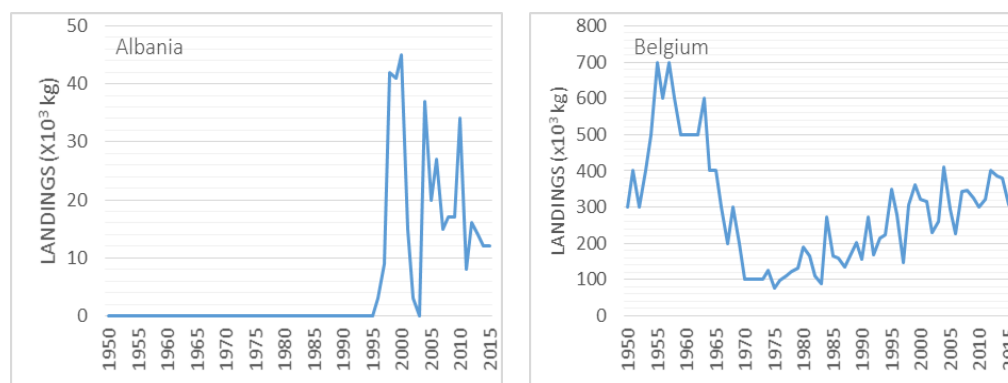


Figure 3.7 (part 1 of 4). Total reported landings ($\times 10^3$ kg) of European flounder (*Platichthys flesus* Linnaeus, 1758) between 1950 and 2015 for each country. Please note that landings of “0 $\times 10^3$ kg” signifies either the absence of landings, landings below 1×10^3 kg, or no data available. The scale of the varies between countries. Data retrieved from FAO (2017).

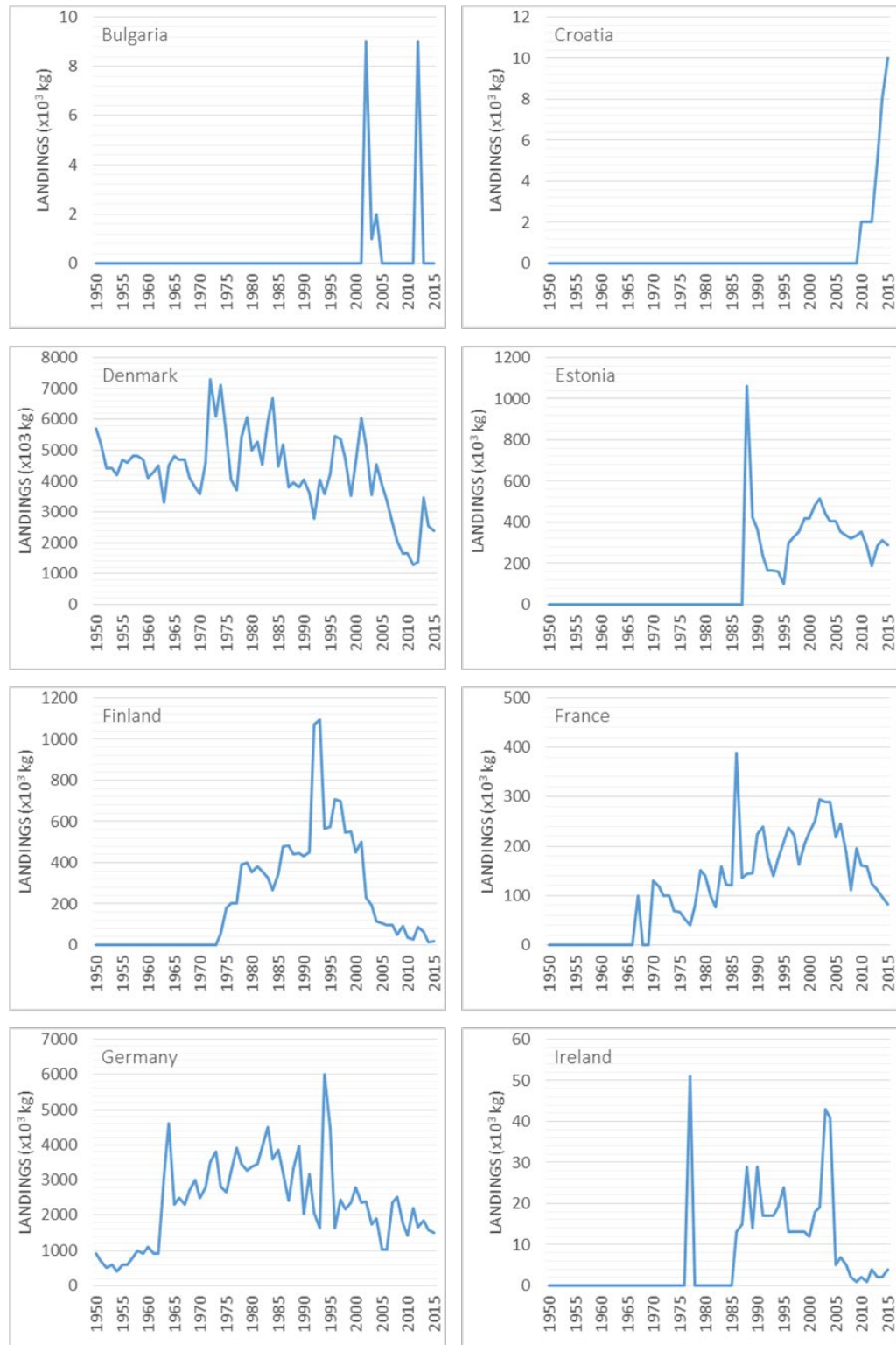


Figure 3.7 (part 2 of 4). Total reported landings ($\times 10^3$ kg) of European flounder (*Platichthys flesus* Linnaeus, 1758) between 1950 and 2015 for each country.

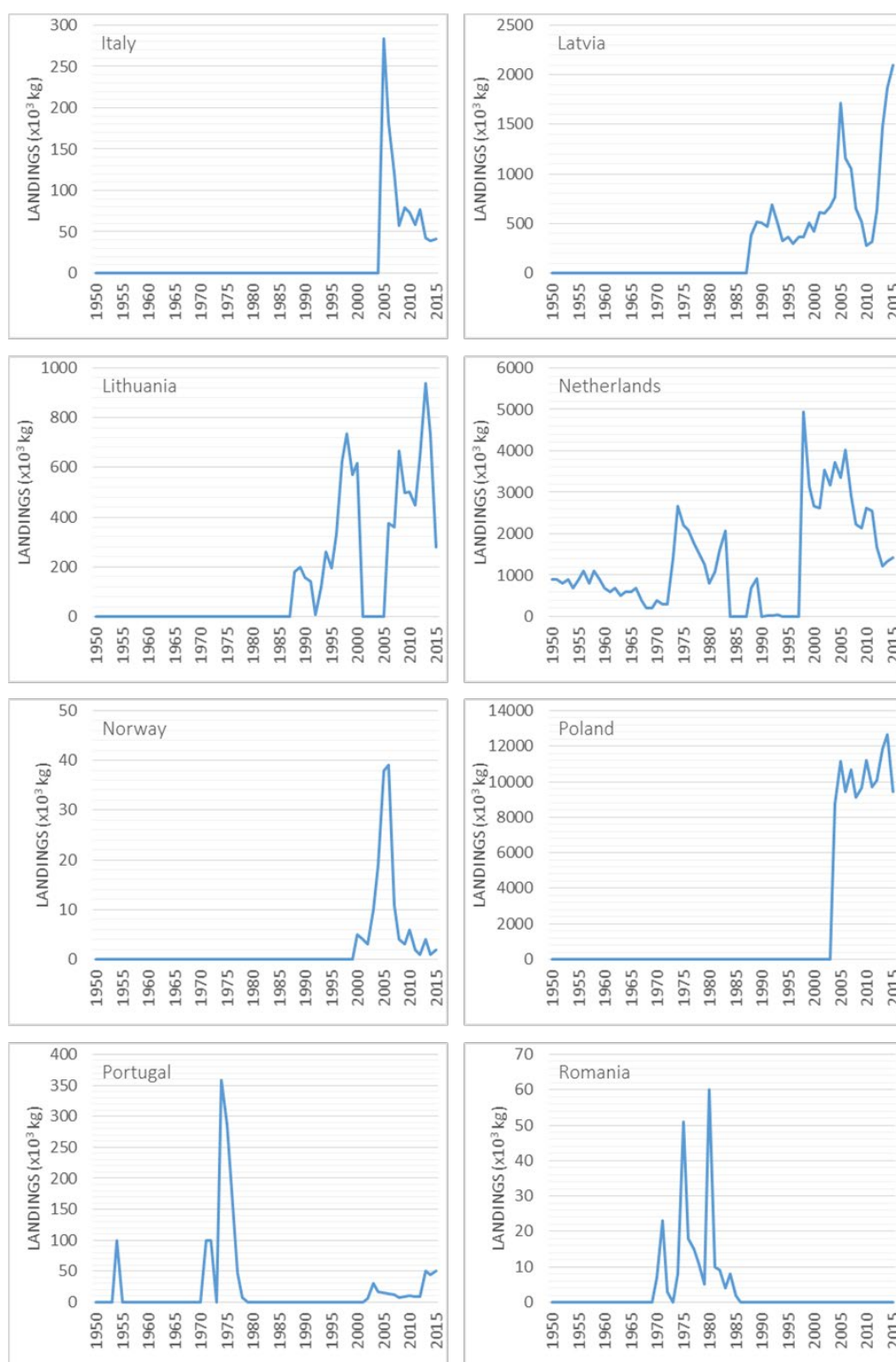


Figure 3.7 (part 3 of 4). Total reported landings ($\times 10^3$ kg) of European flounder (*Platichthys flesus* Linnaeus, 1758) between 1950 and 2015 for each country.

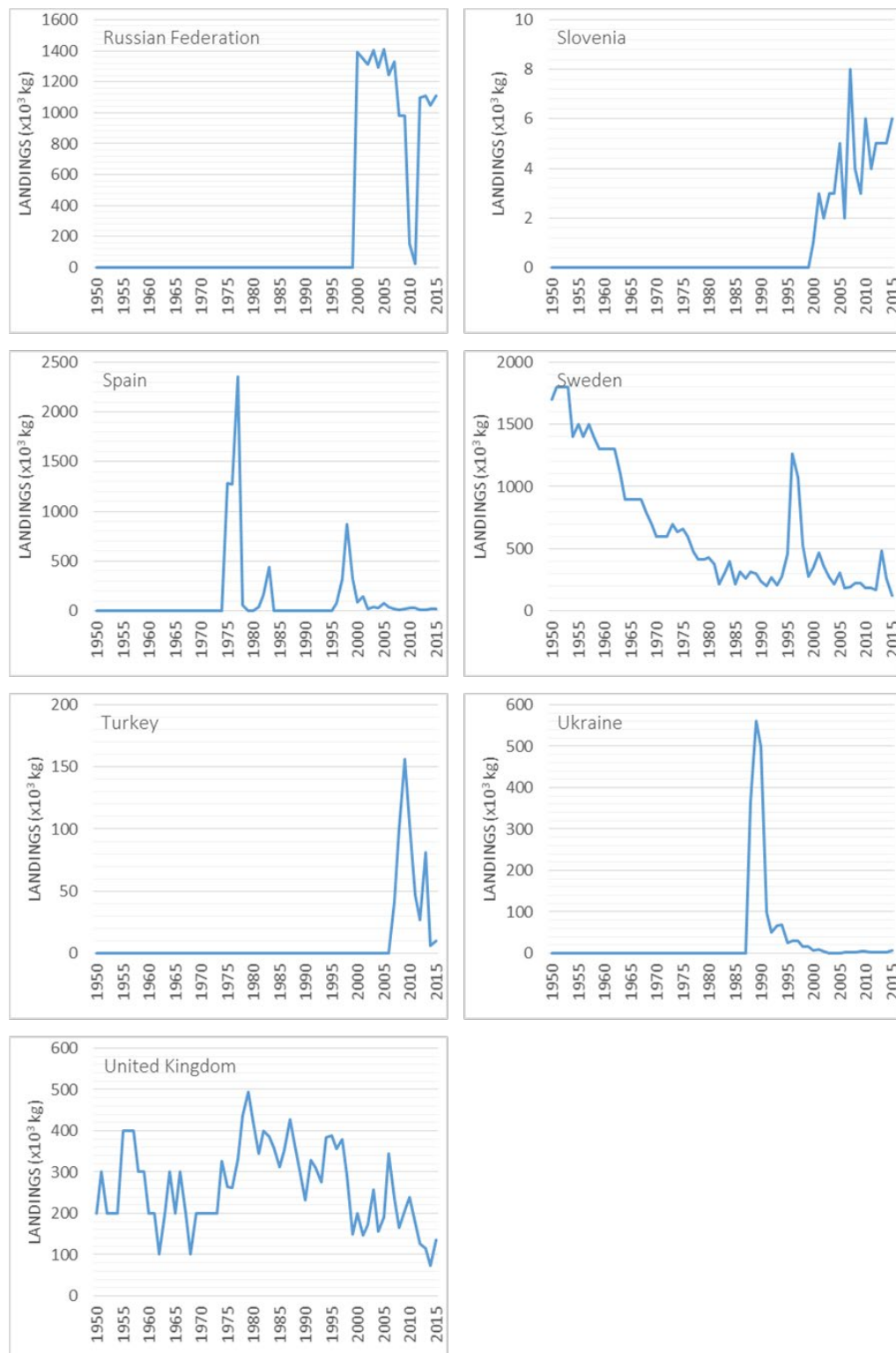


Figure 3.7 (part 4 of 4). Total reported landings ($\times 10^3$ kg) of European flounder (*Platichthys flesus* Linnaeus, 1758) between 1950 and 2015 for each country.

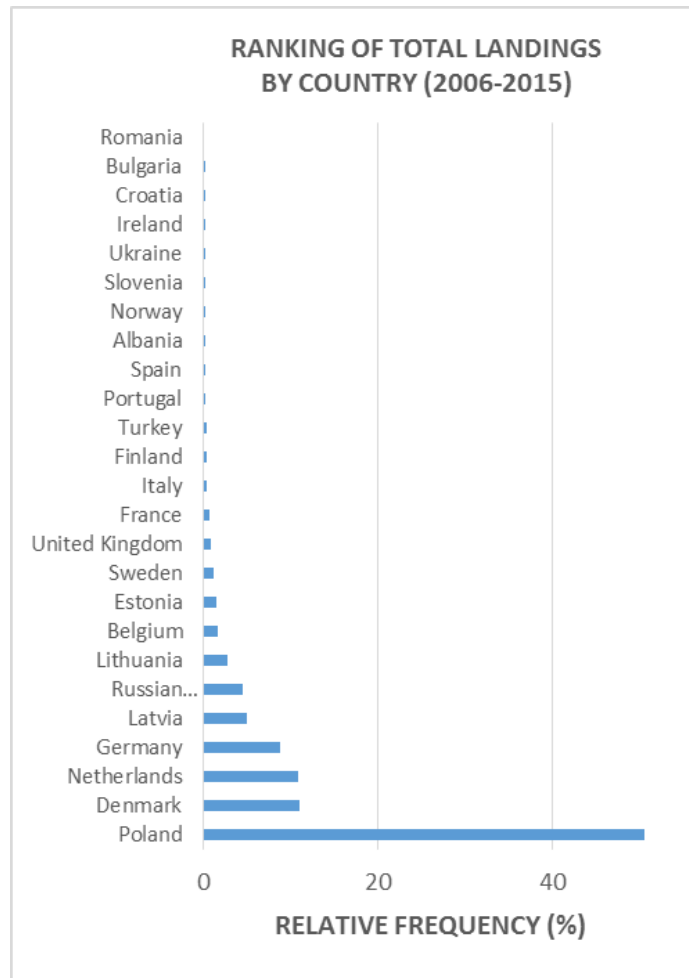


Figure 3.8. Ranking of the relative frequency (%) of the total reported landings (tonnes) of European flounder (*Platichthys flesus* Linnaeus, 1758) by country for the period 2006–2015. Data retrieved from FAO (2017).

The European Union has no specific monitoring programmes for European flounder, and the catch statistics come mostly from official landings. This species has a relatively low economic value. In the Baltic and North seas, flounders are often discarded at sea as bycatch from cod (*Gadus morhua*), and other flatfish fishery (ICES, 2014d). Recreational catches may also have a deleterious impact on flounder populations. Therefore, and as a precautionary measure, flounder monitoring programmes should be implemented to obtain more realistic information on the status of the stock along its distribution.

The decline of some flounder populations has likely been caused by stressors that occurred in the past (e.g. overfishing) or are ongoing (e.g. pollution, and climate change). Therefore, it is recommended that studies focusing on the impact of climate change and other stressors on flounder life history and ecology be implemented in tandem with national monitoring programmes. Only an integrated approach can develop sound policies for the management of European flounder at a broader level. Nonetheless, it is relevant to not neglect fishery in brackish and freshwater habitats, because estuaries and rivers may represent additional reproduction areas for the species (Morais *et al.*, 2011).

3.3 Thin-lipped grey mullet

3.3.1 Taxonomy

Class: Actinopterygii
 Order: Mugiliformes
 Family: Mugilidae
 Scientific name: *Liza ramada* (*Chelon ramada*)

Subspecies, variations, synonyms

Mugil capito
Liza capito
Mugil ramada
Mugil aramada
Liza tamada
Liza alosoides
Mugil dubahra
Mugil maroccensis
Mugil petherici

Taxonomic notes

Mugil ramado, *Mugil cephalus ramado*, and *Liza ramado* have been referred to as synonyms, but are actually misspellings. The spelling “*ramado*” was the vernacular in 1810, and was initially thought to be the correct spelling.

Common names

EN: Grey mullet; DE: Meeräsche; DA: Multe; EE: Kitsashuul-tintkefaal; FI: Ohu-thuulikeltti; FR: Mulet porc; IS: Röndungur; NE: Dunlipharder; NO: Tynnleppet multe; PL: Mugil ramada; RU: Ramada; ES: Mule, Morragute; SV: Tunnläppad multe; PT: Alvor, Tainha, Muge, Fataça.

General characteristics

Maximum length: 70 cm total length (TL)
 Maximum reported weight: 2.9 kg
 Maximum reported age: 10 years (Froese and Pauly, 2015)
 Generation time: 3 years (Mediterranean region)

Portugal: Maximum TL in the Tagus Estuary (specimens sampled in spring and summer of 2007): 45.5 cm (male), 44.4 cm (female); total weight (TW) of the 45.5 cm TL male was 783.5 g; TW of the 44.4 cm female was 718.23 g (S. Pedro, pers. obs.); generation time: 4–5 years.

Ireland: Maximum length: 61 cm TL; maximum published weight: 2.86 kg (ISFC, 2015).

The thin-lipped mullet is a catadromous pelagic shoaling species (Rafalah and El-Mor, 2014). It is a euryhaline species, and is widely distributed in the Mediterranean Sea. Its range in coastal waters of the Northeast Atlantic extends northward to the UK (except northern parts of Scotland), the North Sea, and the southern part of the Baltic, to Morocco in south. It mainly appears in summer in the northern parts of the range (Marine

Species Identification Portal, 2016). Turan (2015) postulated a more extensive distribution, which includes the full extent of the Norwegian coastline and down to Mauritania on the African coast. The thin-lipped mullet is one of the most abundant species of Mediterranean mullets. However, it is rare north of the English Channel, with the exception of south of Ireland (Maitland and Herdson, 2009). It is one of three mugilid species recorded in Ireland, where it has a limited distribution, is not commonly recorded, and was first documented in 1971 (Kennedy and Fitzmaurice, 1972).

The thin-lipped mullet is principally an inshore species. Adult thin-lipped mullet frequent estuaries and coastal lagoons, and penetrate upstream into the lower freshwater reaches of rivers (Almeida *et al.*, 1995). *Liza ramada* dominates mugilid biomass in salt-marsh habitat in northern France (Laffaille *et al.*, 2000), and in the Loire River, adults have been recorded in freshwater up to 350 km upstream of tidal influence (Sauriau *et al.*, 1994). In Portugal (mainland), it is one of five mugilid species, and is present in all the main estuarine systems from north to south (França *et al.*, 2011). *Liza ramada* is not an important economic halieutic resource in Portugal, despite its abundance, widespread occurrence, and exploitation in many Mediterranean countries (Oliveira and Ferreira, 1997). There is some interest in fishing this species by local fishers in the Tagus Estuary, but no commercial exploitation. Specialist anglers also target thin-lipped mullet along the south coast of Ireland, but it is not of any commercial interest. In northern Europe, thin-lipped mullet is the most abundant grey mullet species occurring in freshwater (Maitland and Herdson, 2009), and it is known to tolerate fluctuations in salinity and variations in water quality (Lasserre and Gallis, 1975; Thomson, 1990; Koutrakis, 2004; Cardona, 2006). Juveniles inhabit both the littoral zone and brackish waters, but prefer mesohaline sites (≤ 15 salinity) (Cardona, 2006). Thin-lipped mullet fry (< 30 mm TL) are mainly zooplanktivores. Larger fry adopt a mainly benthivorous diet (dominated by detritus, benthic organisms, epiphytic algae, plankton, and pelagic larvae; Kottelat and Freyhof, 2007), and are often opportunistic feeders (Bartulovic *et al.*, 2007). This varied feeding strategy continues over its adult life (Cardona, 2015).

Global production of Mugilidae was ca. 700 000 t in 2013, with 80% attributed to commercial fishing and the remainder to aquaculture (Crosetti, 2015). Production statistics for mullets tend to be reported using generic terms, with little species-specific identification. However, in Europe, a thin-lipped mullet commercial catch of 175 t was reported in 2013, all attributed to France (Crosetti, 2015). The capture of wild mullet fry, including *L. ramada*, to support commercial aquaculture remains a common practice in the Mediterranean (Sadek and Mires, 2000), as artificial propagation of mullet fry is not widely practiced owing to the high costs involved (Crosetti, 2015).

3.3.2 Life cycle and migrations

Adult thin-lipped grey mullet migrate in shoals from lakes, rivers, and coastal lagoons to offshore spawning areas. Females generally mature at age three (Campillo, 1992), whereas males can mature at two (Kottelat and Freyhof, 2007). Length-at-maturity averages 26 cm, and ranges from 25 to 32 cm (Binohlan, 2000). Portuguese populations in the Tagus Estuary were described as having females maturing at ages 4–5, with an average length 21–25 cm (Almeida, 1989). In the Mira Estuary, both sexes mature at age four, with an average length of 18 cm (Almeida, 1996). Populations tend to be female-dominated (Ergene, 2000; El-Halfawy *et al.*, 2007; Glamuzina *et al.*, 2007), and males tend to mature at a smaller total length than females (El-Halfawy *et al.*, 2007). Spawning typically occurs between September and February, across the species distribution (Maitland and Campbell, 1992; Almeida, 1996; Ergene, 2000; Glamuzina *et al.*,

2007; El-Halfawy *et al.*, 2007). Claridge and Potter (1985) deduced that the Bristol Channel population spawns between April and June, which may be related to environmental factors in this northerly part of the species' range. Portuguese populations also show some time variations in their spawning migration. In the Mondego River estuary, a fish-pass monitoring study recorded a maximum number of adults migrating downstream during August and September (more than 350 000 in 2013 and more than 450 000 in 2014; Almeida *et al.*, 2015). In contrast, the spawning migration is estimated to occur between September and November in the Tagus River estuary (Almeida, 1989), and between November and February in the Mira River estuary (Almeida, 1996).

As with most mugilids, fertilized eggs develop at sea, and motile larvae undertake a trophic migration shoreward (Koutrakis *et al.*, 1994) to nursery waters in lagoons, rivers and accessible lake habitats (Thomson, 1966). Larvae use selective tidal stream transport to aid migration within and through estuaries (Trancart *et al.*, 2012) to reach nursery areas. Young-of-the-year start entering the Portuguese estuaries in late winter to early spring (Salgado *et al.*, 2004; Almeida, 1989). In the Douro Estuary, in northern Portugal, the minimum reported SL (Standard Length) of larvae entering estuarine waters was 2.0 cm (average SL 3.6 cm) (Figueiredo, 2003). In the Mondego Estuary, in the central region of Portugal, the first larvae reach estuarine waters around April, with a minimum length of 2.7 cm (Ramos, 2001). During spring, there is an upstream migration of adults and some older juveniles, which is likely for trophic purposes. By the end of this period, the fish present a peak in the hepatosomatic index (HIS), consistent with the need to build up energy reserves for the upcoming spawning migration. In the Mondego, the upstream migration starts during April, is significantly higher in June and July, and lasts until September/October (Almeida *et al.*, 2015). In the Tagus estuary this trophic migration occurs between mid-April and June (Almeida 1989). In this estuarine system, saltwater taxa were found in the stomach contents of grey mullets captured in freshwater environments, providing evidence for a movement from saline to freshwater in a short period of time (Almeida, 1989).

3.3.3 Stock structure and population dynamics

Thin-lipped mullet populations from the Mediterranean and the Adriatic coastal regions typically comprise individuals of ages 0–8 years old (Quignard and Authem, 1981; Almeida *et al.*, 1995; Kraïem *et al.*, 2001; Glamuzina *et al.*, 2007). In tropical Atlantic waters, a 10-year lifespan has been reported (Thomson, 1990).

Baltic Sea (Germany, Sweden, and Denmark)

Mullet species are of minor commercial importance. No regular scientific sampling occurs (Uwe Krumme, pers. comm.) because annual landings by any Baltic member state are well below 200 t (the threshold below which a country does not have to sample commercial catches within the scope of the data collection framework (DCF) of the EU). Total mullet catch between 2009 and 2012 was 45.8 t, and comprised *Chelon labrosus* (4.39%), *Mugil cephalus* (14.57%), and Mugilidae sp. (81.04%) (ICES database Fish-Frame/RDB). Annual catches were consistent over this period, ranging from 9.1 to 13.4 t. The highest fraction of landings in the Baltic Sea were from Denmark (79.55%), followed by Germany (14.57%) and Sweden (5.88%). Mullet species are regularly landed in summer, but no species identification is undertaken on infrequently recorded species, especially juveniles.

On the west coast of Denmark, thin-lipped mullet are reported as a rare visitor. *C. labrosus* is the most commonly occurring mugilid species, but rarer mullet species may

not be recorded owing to a lack of awareness of the likelihood of different species occurring.

Belgium

Liza ramada occurs in two estuaries, the IJzer and the Zeeschelde. The IJzer Estuary is a short (4 km long), polyhaline estuary, which has been monitored since 2008 using fykenets. On each sampling occasion, thin-lipped mullet were caught together with *Liza aurata*. One major issue with the IJzer is the presence of a complex sluice system which only allows mullets to pass farther upstream on rare occasions. Although regularly caught, this species does not belong to the top 10 most commonly caught species in Belgium. There is no real threat to the species. *Liza ramada* is subject to professional fishing in the Netherlands, but not in Belgium.

Ireland

In general, mullet species are of minor commercial importance. Landings of undifferentiated mullet species (likely to be *C. labrosus*) between 2003 and 2012 averaged 3.65 t (1.1–6.9 t) (SFPA IFIS). Thin-lipped mullet are a recreational catch-and-release angling species, but are not heavily targeted on account of its limited distribution in Irish waters, being mainly confined to the south coast. No records of thin-lipped mullet in Irish waters are documented in the ICES DATRAS database.

Poland

Thin-lipped mullet have been observed occasionally. The first record of *Liza ramada* came from near Stepnica Harbour on 19 June 2008 (caught using fykenets; Panicz and Keszka, 2015). The second specimen, captured in October 2015 at the same site, was 25 cm TL and weighed 80 g (Panicz and Keszka, pers. comm.). Previously, only *Chelon labrosus* was known to occur in the Odra Estuary, from a specimen captured using a fykenet in the northern part of Lake Dąbie (Odra Estuary) on 14 November 2007 (Czernejewski *et al.*, 2008). It is assumed that *M. cephalus* abundance is low, but it is well known to fishers and observations are regular.

Portugal

Liza ramada is one of the five mugilids occurring in Portuguese watersheds and is present in all the main estuarine systems from north to south (França *et al.*, 2011). High abundance and biomass are observed in the areas of occurrence. However, population trends and estimates are not available, as it has little commercial interest. Its abundance has been described for a few Portuguese water bodies. For example, Tagus Estuary surveys showed *L. ramada* constitutes up to 88% of the nekton biomass of saltmarsh creeks (Salgado *et al.*, 2004), and is by far the most abundant mugilid species (Almeida, 1989). In the Mondego River, 90–99% of the fish using a fish pass during 2013–2014 belonged to this species (ca. 1 million per year) (Almeida *et al.*, 2015). In the Guadiana Estuary, southern Portugal, monthly surveys during 2000–2001 in Castro Marim saltmarsh showed 15.9% of the fish biomass belonged to this species (Veiga *et al.*, 2006). In terms of mean density (number of individuals 1000 m⁻²), the Mondego Estuary presents the highest value (5.4), followed by the Mira (0.4) and Sado (0.2) estuaries. Regarding mean biomass (g 1000 m⁻²), the Mondego (172.5), Tagus (58.8), Guadiana (42.6), and Sado (15.1) estuaries have the highest thin-lipped grey mullet biomass of the Portuguese systems (França *et al.*, 2011). Data from 1986 refer to 15 t of mugilid species out of a total of 25 t of fish captured (Almeida *et al.*, 1992). The Portuguese populations of *Liza ramada* are considered to represent up to 4% of the global population of this species (Cabral *et al.*, 2005).

3.3.4 Genetics

The Mugilidae family comprises 20 genera and seven major lineages (Durand *et al.*, 2012). Their phylogeny and taxonomy is currently undergoing revision on account of recent advances in genetic and morphometric techniques (Durand *et al.*, 2012, Gonzalez-Castro and Ghasemzadeh, 2015; Rossi *et al.*, 2015). Morphologically, members of the Mugilidae family are similar, and five species (*Liza ramada*, *L. aurata*, *L. saliens*, *Chelon labrosus*, and *Mugil cephalus*) can inhabit the same general habitat range. Papa *et al.* (2003) demonstrated that only genetic analysis could discriminate smaller individuals of *L. ramada* from *L. saliens*. Several recent studies (Rossi *et al.*, 2004; Heras *et al.*, 2009; Turan *et al.*, 2011; Nematzaheh *et al.*, 2013; Panicz and Keska, 2016) have referenced *L. ramada*, and concluded that a *Liza* family monophyletic grouping is not supported. Morphometrically, *Liza* is shown to cluster closest to *Chelon* (Turan *et al.*, 2011) and, more specifically, using genetic analysis, *L. ramada* clustered to *C. labrosus* (Durand *et al.*, 2012; Panicz and Keska, 2016).

3.3.5 International status with country highlights

OSPAR Convention:	Not Listed
EC Habitats Directive:	Not listed
CITES:	Not Listed
Bern Convention:	Not Listed
Bonn Convention:	Not Listed
IUCN Criteria:	LC (Freyhof 2018)
Global IUCN Red List Category:	LC (Freyhof 2018)
European Red List Category:	Not Listed
HELCOM Red List:	Not Listed

3.3.6 Distribution summary

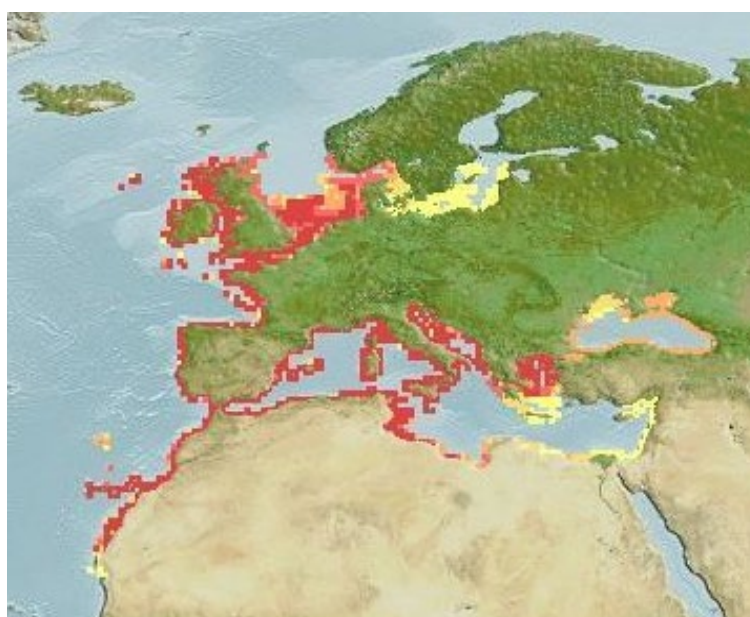


Figure 3.9. Range of thin-lipped grey mullet (*Liza ramada*) (Froese and Pauly, 2016).

3.3.7 Country-by-country changes in distribution

The thin-lipped grey mullet appears to have been extending its range over recent decades. Jonsson and Jonsson (2008) have documented the first thin-lipped grey mullet in southern Norway. In Ireland, sporadic occurrences of juveniles and adults have been documented since the 1970s (Kennedy and Fitzmaurice, 1972; Walsh, 1993; Jovanovic *et al.*, 2007). In recent years, specimen-hunter anglers have been recording small numbers of thin-lipped grey mullet (ISFC, 2015), mainly between June and September, in estuaries or adjacent to lagoons along the south coast of Ireland. Juveniles have been recorded sporadically in some inshore waters on the east and south coasts of Ireland since the mid-2000s (IFI, unpub.). According to Quigley (2012), thin-lipped grey mullet have been recorded in UK and Scandinavian waters for more than 200 years, and may also have been present in Irish waters over this period. In his review, Turan (2015) documents *L. ramada* in Icelandic waters, but there are no records of this species being caught in Icelandic waters (Jonbjorn Palsson, pers. comm.).

3.3.8 Threats

The same factors that threaten coastal waters, threaten the success of thin-lipped grey mullet, including eutrophication and changes to hydrology and salinity. Although thought to be more resilient to eutrophication than other mullet species (Crosetti and Cataudella, 1995), there is evidence that *L. ramada* inhabiting eutrophic conditions can develop intersex gonads and have reduced fecundity (Tancioni *et al.*, 2014). Ortiz-Zaragoza *et al.* (2014) suggest that mullet may act as bio-indicators for endocrine-disrupting compounds in transitional waters. The toxic organic pollutant polychlorinated biphenyl has been detected in thin-lipped grey mullet muscle tissue in French estuaries (Bocquené, 2013). Potentially toxic elements, such as Hg and Ni, were observed at levels above sediment quality guidelines in the stomach contents of thin-lipped grey mullets captured in the Tagus Estuary, Portugal (with sediment representing a large proportion of stomach contents; Pedro, 2015). This is likely a consequence of metal hotspots, which are still observed in the estuary (e.g. Canário *et al.*, 2007). The feeding habitats of *L. ramada*, potentially exposes them to contaminated sediments (Pedro *et al.*, 2008). Plankton blooms from nutrient inputs have coincided with a thin-lipped grey mullet stock collapse in a major Sardinian lagoon since 1999 (Murenu *et al.*, 2004).

Mullet fry overfishing for aquaculture could threaten wild stocks, although there are divergent opinions on this (Crosetti, 2015). Bird predation is also an increasing threat to coastal fish species, especially in semi-enclosed estuaries and lagoons (Cataudella *et al.*, 2015). Growing populations of the great cormorant (*Phalacrocorax carbo*) pose a substantial threat to fish life in coastal lagoons and estuaries. Population estimates for central Europe suggest that the great cormorant population has grown 20-fold in 25 years, reaching 1.8 million birds that consume up to 300 000 t of fish from European waters annually (Kindermann, 2008).

Climate change is also a potential hazard to *L. ramada*, as it is likely to affect water temperature, salinity, sea levels, erosion, air temperature, and UV radiation levels. These factors, when combined, could contribute to the degradation of critical habitats frequented by shoaling mullet, such as sea grass meadows (Cataudella *et al.*, 2015).

3.3.9 Conclusion – future

Increasing sea temperatures are likely to increase the distribution range of thin-lipped grey mullet. To accurately document changes in the distribution or status of stocks, greater emphasis must be placed on improving species-level identification of all mullet species. Eutrophication poses a threat to thin-lipped grey mullet, and pollutants and

inputs to rivers, lagoons, and transitional waters must be reduced. Within the EU, the target of restoration of good ecological status (GES) to waterbodies under the Water Framework Directive (WFD) should contribute to protection of all species/habitats. As the current status and populations trends of thin-lipped mullet stocks are poorly understood, monitoring programmes should be instigated. Monitoring of fish populations in transitional and adjacent waters under the WFD will provide regular biodiversity reports at intervals, but targeted annual programmes may also be required to provide robust stock status time-series data in countries where the species is well established. All such monitoring programmes will contribute toward increasing knowledge of the life-history characteristics and ecology of thin-lipped mullet in addition to supporting any management actions that may be required for the species.

4 Species with totally or partially anadromous populations

4.1 Burbot

4.1.1 Taxonomy

Class: Actinopterygii

Order: Gadiformes

Family: Lotidae

Scientific name: *Lota lota*

Subspecies, variations, synonyms

Gadus lota, *Enchelyopus lota*, *Lota lota lota*, *Molva lota*, *Gadus lacustris*, *Lota lota lacustris*, *Gadus maculosus*, *Lota lota maculosa*, *Lota maculosa*, *Molva maculosa*, *Gadus maculosa*, *Gadus compressus*, *Lota compressa*, *Lota vulgaris*, *Lota fluviatilis*, *Lota marmorata*, *Lota inornata*, *Lota brosmia*, *Lota communis*, *Lota linnei*, *Lota vulgaris obensis*, *Lota lota kamensis*, *Lota lota leptura*, *Lota lota onegensis*, *Lota lota asiatica*

Taxonomic notes

None

Common names

DE: Quappe; DA: Kvabbe; ET: Luts; FI: Made; LV: Vēdzele; LT: Vėgėlė; PL: Miętus; RU: Налим; SV: Lake.

General characteristics

Maximum total length: 150 cm

Maximum weight: 34 kg

Maximum age: 25 years

Generation time: 8.3 years

Burbot is the only freshwater species in the cod family, and one of the few freshwater fish species with a Holarctic distribution. Several subspecies have been recognized, but no consensus on their taxonomy exists (see van Houdt *et al.*, 2003 and references therein). Although mostly regarded as a freshwater species, burbot can also withstand the low salinities of brackish seas, and it performs anadromous migrations to these waters. It is commercially and recreationally important in some regions, mostly in Eurasia (Stapanian *et al.*, 2010).

4.1.2 Life cycle and migrations

Burbot is a cold-water species and lives mostly in freshwater, or in coastal waters of brackish seas. Burbot spawn in winter (December–February), and young-of-the-year hatch in spring (April–May). According to Lehtonen (1998), two different migratory phenotypes exist along the Baltic coast: (i) individuals that spawn at sea and spend their whole life in coastal waters, and (ii) individuals that spawn in freshwater (rivers, streams, lakes), but otherwise live at sea. However, this notion does not seem to apply to the entire Baltic Sea. Rohtla *et al.* (2014) demonstrated that 96% of the burbot sampled in the Väinameri Sea (Estonia) were of freshwater origin; and even the remaining 4% were probably spawned in the lower reaches of rivers and drifted to the estuaries as egg/larvae. Interestingly, HELCOM (2013b; based on HELCOM, 2012a) suggests that

brackish-water spawning is widespread in the Baltic Sea. However, Rohtla *et al.* (2014) is the only reference with quantitative data on burbot spawning habitat preferences. Other references are based on observational data or assumptions and, therefore, should be regarded as inconclusive. It might also be that coastal areas of Estonia are currently not suitable for burbot spawning, as has also been suggested for most of the pike (*Esox lucius*) inhabiting the Väinameri Sea (Rohtla, 2015).

The progeny of anadromous burbot inhabit freshwater for 2–12 months prior to descending to the sea (Eriksson and Müller, 1982; Rohtla *et al.*, 2014). Burbot migrations in the Baltic Sea can exceed 20 km, but are mostly within 10 km of the natal river (Hudd and Lehtonen, 1987). Natal homing is well developed in burbot (Hedin, 1983; Hudd and Lehtonen, 1987). Burbot living in the Baltic Sea usually reach sexual maturity at the age of 2–3 years (Lehtonen, 1998; Rohtla *et al.*, 2014).

4.1.3 Stock structure and population dynamics

No population estimates are available for coastal burbot in the Baltic Sea, but populations are most likely declining. Swedish coastal landings have decreased from 4 t in 1999 to 1 t in recent years. Landings in Finland have also decreased from an average of 130 t in the 1980s to an average of 62 t after 2000. However, to some extent, this is owing to lower effort. In the Väinameri area (the best fishing area for burbot in Estonian coastal water), annual commercial catches have declined to ca. 1 t per year. However, in the Lithuanian part of the Curonian Lagoon (almost entirely freshwater), there has been an increasing trend in commercial landings over the last 20 years, with a maximum of almost 25 t in 2015 (41 t for the whole lagoon including the Russian part). In contrast, commercial landings during the last 10 years in the Lagoon (including both Lithuanian and Russian parts) are more than threefold lower than landings in the 1980s. Restocking is a common practice in Lithuania, in the delta of the Nemunas River; but the contribution of the restocked burbot to the stock has never been estimated and remains unknown (L. Lozys, pers. comm.).

4.1.4 Genetics

Phylogenetic studies have demonstrated that two distinct forms of burbot exist worldwide: one in North America, south of the Great Slave Lake (*Lota lota maculosa*), and one in Eurasia and the remainder of the Nearctic region (*Lota lota lota*) (van Houdt *et al.*, 2003). No genetic data are available for anadromous burbot.

4.1.5 International status with country highlights

International

OSPAR Convention:	Not listed
EC Habitats Directive:	Not listed
CITES:	Not listed
Bern Convention:	Not listed
Bonn Convention:	Not listed
IUCN Criteria:	A2b
Global IUCN Red List Category:	LC
European IUCN Red List Category:	LC

HELCOM Red List: NT

National IUCN Red List classifications

Denmark:	– / LC
Estonia:	– / LC
Finland:	– / LC
Germany:	Minimum landing size (30 cm) / LC
Latvia:	National angling rules / –
Lithuania:	Minimum landing size (49 cm for commercial, and 45 cm for recreational fishers), bag limits and closed season for recreational fishers / –
Poland:	Minimal landing size (40 cm for commercial fishery , and 25 cm for recreational anglers); fishing is not allowed from 1 December till 28/29 February / –
Russian Federation:	– / –
Sweden	– / NT

4.1.6 Distribution summary

Facultatively anadromous populations (Figure 4.1): Denmark, Estonia, Finland, Germany (rare), Latvia (rare); Lithuania (rare; but abundant in the Curonian lagoon which is almost entirely freshwater); Poland (not confirmed clearly by scientific research; the species is present in the Polish part of Vistula Lagoon and Szczecin Lagoon), the Russian Federation, Sweden, and western Caspian Sea countries (Table 4.1).



Figure 4.1. Distribution of burbot (NatureServe, 2013b).

Table 4.1. Distribution of anadromous burbot in the Baltic Sea by country.

Country	Distribution
Denmark	Data not available.
Estonia	Common in the Väinameri Sea region and around Saaremaa and Hiiumaa islands. Most abundant runs of anadromous burbot occur in rivers/streams that disembody to Matsalu and Saunja bays (e.g. Kasari, Rannamõisa, Tuudi, Penijõgi, Liivi, Taebala, Salajõgi rivers). Larger runs probably also occur in Paadrema, Põduste, and Võlupe rivers, and in the Kõrgessaare region (Hiiumaa). Rare in coastal waters of northern and southwestern Estonia.
Finland	Common throughout the coastal sea.
Germany	Rare in coastal waters. The only significant burbot population is found in the Oder Estuary (Stettin Lagoon) (H. Winkler, pers. comm.).
Latvia	Rare in coastal waters.
Lithuania	Common in the eutrophic Curonian Lagoon. They do not migrate to brackish coastal waters. For spawning, they migrate upstream to the Nemunas River (L. Lozys, pers. comm.).
Poland	Present in Polish part of Vistula Lagoon and Szczecin Lagoon
Russia	Data not available.
Sweden	Not known.

4.1.7 Country-by-country changes in distribution

Despite low fishing pressure, the numbers of burbot are declining worldwide (Stapanian *et al.*, 2010). The burbot status in the following countries was obtained from Stapanian *et al.* (2010), but probably applies mostly for freshwater resident burbot:

Denmark:	Not ranked/unknown
Estonia:	Declining
Finland:	Declining
Germany:	Stable (rare in coastal waters)
Latvia:	Stable (rare in coastal waters, numbers in the rivers increased)
Lithuania:	Stable
Poland:	Vulnerable
Russian Federation:	Stable
Sweden	Declining

4.1.8 Threats

Past and current threats (Habitats Directive Article 17 codes): Eutrophication (H01.05), Climate change (M01.01, M01.04).

Future threats (Habitats Directive Article 17 codes): Eutrophication (H01.05), Climate change (M01.01, M01.04).

The problems associated with Baltic Sea burbot generally concern freshwater recruitment, as many spawning rivers have been polluted, acidified, or the habitat has undergone change (Hudd *et al.*, 1983; Stapanian *et al.*, 2010). In addition, the deterioration of

coastal spawning grounds may be an important factor affecting brackish-water spawning burbot. In Estonia, and possibly also in Sweden, the burbot decline in the coastal sea is probably caused by predation by the increasing numbers of cormorants (*Phalacrocorax carbo*; Eschbaum *et al.*, 2003; J. Olsson, pers. comm.), and overfishing, resulting from a lack of any fishing regulations in the sea or the spawning rivers. It is estimated that cormorants in the Väinameri Sea consume ca. 100 t of burbot annually, whereas the commercial catch in that region is only 1–2 t per year (Eschbaum *et al.*, 2003). However, recreational fishers in Estonia and Sweden most likely land somewhat more than commercial fishers.

Due to climate warming, burbot faces new challenges in the southern limits of its distribution. However, it seems that in the temperate regions (e.g. in the Baltic Sea), increases in annual temperatures result in milder winters and colder summers. For burbot, this means favourable growth conditions throughout the year; and high growth rates have indeed been recently reported from the Baltic Sea (Rohtla *et al.*, 2014). However, the effect of milder winters on burbot spawning is currently unknown.

The monitoring programmes for anadromous burbot are listed in Table 4.2.

Table 4.2. Monitoring programmes and data for anadromous burbot in the Baltic Sea countries.

Country	Monitoring and data
Estonia	Monitored annually in several sea areas as part of the national coastal sea fish monitoring programme (although rarely observed in summer surveys); only occasional data from electrofishing surveys in rivers; official landings from the sea and coastal rivers.
Denmark	Data not available
Finland	Data not available
Germany	Rare in coastal waters. No special monitoring for burbot; there are electrofishing surveys in the most important streams and rivers (EU-Water Frame Directive) and catch statistics from commercial fishery.
Latvia	Rare in coastal waters. Official landings from the rivers; monitoring in the rivers.
Lithuania	There are official landings available from the Curonian Lagoon (commercial fishery). There is no cold-water monitoring carried out for burbot stock assessment.
Poland	Piecemeal data available from Vistula and Szczecin Lagoon as a part of the monitoring conducted within the Multiannual Programme for Collection of Fisheries Data.
Russian Federation	Data not available
Sweden	Catch per unit effort from commercial coastal fishery

4.1.9 Conclusion – future

Management recommendations:

- Monitoring programmes specially designed for targeting burbot should be initiated to obtain information about stock status. This should involve gillnet surveys in the sea during colder months when burbot become active, and electrofishing surveys in coastal rivers.

- To maintain/increase abundance in stable-state stocks, or to stop the decline in declining stocks, fishing regulations should be implemented. Totally closed spawning season and minimum landing size should be established.
- To enhance natural reproduction of anadromous burbot, channelized rivers should be restored to naturally meandering rivers.
- Restocking is recommended, but only using local broodstock.
- As the reasons for declining stocks are not well known, it is necessary to increase knowledge of the species life history and ecology, in order to suggest meaningful action plans.

4.2 Coastal grayling

4.2.1 Taxonomy

Class: Actinopterygii
Order: Salmoniformes
Family: Salmonidae
Scientific name: *Thymallus thymallus*

Subspecies, variations, synonyms

Salmo thymallus

Taxonomic notes

None

Common names

DE: Äsche; DA: Stalling; EE: Harjus; FI: Harjus; LV: Alata; LT: Kiršlys; PL: Lipień;
RU: Evropeiskiy kharius; SV: Harr.

General characteristics

Length: 20–65 cm
Weight: 0.2–3 kg
Maximum age: 30 years
Generation length: 6–7 years

Grayling is a widely spread species within the Baltic Sea drainage area, particularly in high altitude streams. Grayling also inhabits the brackish water areas of the northern Baltic Sea, where it mainly occurs in the coastal areas of the Bothnian Bay. Two different ecotypes of grayling are recognized in the coastal areas of the Baltic Sea: a sea-spawning ecotype, and a migratory ecotype reproducing in rivers. Both ecotypes occur only sporadically in the coastal areas of Sweden and Finland, and have not been observed south of the Gulf of Bothnia. Both sea-spawning and anadromous grayling are a rather unique phenomenon that, to our knowledge, do not occur anywhere else in the world.

Both ecotypes of coastal grayling are currently not important for coastal recreational and/or commercial fishery because of their weak stocks and conservation status. However, they have a high value in a nature conservation context. Given an improved stock status, the coastal grayling could have a large potential for recreational fishery.

4.2.2 Life cycle and migrations

Like other salmonids, coastal grayling demands clear, cold, well-oxygenated, and fast-flowing water. It prefers habitats with non-vegetated stony or gravel bottoms. In addition to stream habitats, it also occurs in oligotrophic lakes and the northernmost part of the Baltic Sea. In lakes and coastal areas, grayling occupies shallow areas, with stones mixed with gravel, that are exposed to wind and waves. Grayling can tolerate brackish water with salinities less than 5.5 ppt (Ehnholm, 1937; Müller and Karlsson, 1983), and grayling eggs and larvae have been observed in salinities of 3–3.3 ppt in coastal areas (Hudd *et al.*, 2006; Broman, pers. comm.). Coastal migratory grayling ascend rivers to spawn early in spring after ice break-up, whereas the sea-spawning ecotype reproduces in coastal areas during the same period (Johnson, 1982). The spawning habitat

in rivers consists of stones and gravel without vegetation or other organic, oxygen-consuming material (Gum, 2007). Directly after spawning, the migratory grayling descends back to coastal areas, where it favours exposed outer shores, mostly in shallow waters (Johnson, 1982). The migrations of coastal grayling are not well known. Marking experiments with a limited number of fish (Natural Resource Institute Finland, unpub.) have indicated that the sea-spawning ecotype is quite stationary, with migration distances not exceeding 30 km. Most of the tag returns were located < 10 km from the release site.

Old data on coastal grayling food preference (Ehnholm, 1937) indicates that grayling feed mostly on amphipod crustaceans, molluscs, various insects, and fish fry. An unpublished study by Sandström (1996) showed that the summer diet of 45 coastal graylings (12–45 cm TL) mainly consisted of chironomids, flying insects, amphipod crustaceans, and tricopters. Data on the growth of coastal grayling is sparse, but a length of 35 cm is often reached by age four (reviewed in Keränen, 2014, unpub.), when fish have also reached maturity.

4.2.3 Stock structure and population dynamics

As described earlier, there are two distinct ecotypes: one anadromous and one sea-spawning. No population estimates are available for coastal grayling populations, thus it is hard to analyse demographic variation and make detailed studies on population dynamics.

4.2.4 Genetics

The genetics of coastal grayling have not been thoroughly documented. The few studies that exist indicate that the majority of local populations are classified as belonging to the eastern Baltic grayling lineage, which also consists of inland stocks (Swatdipong, 2009). These populations with an Eastern origin are mainly composed of the river-spawning grayling stocks in Finland, and the sea-spawning stocks in Finland and Sweden. In Swedish coastal areas, the anadromous grayling that spawns near the mouths of rivers appear to have a mixed origin, and is more closely related to the western Baltic lineage of grayling than the sea-spawning variety (Alanärä *et al.*, 2006; Nilsson and Alanärä, 2006; Alanärä, 2008). There has been some mixing of stocks connected to occasional reproductive migration events of fish (Koskinen *et al.*, 2000). However, stocks are still considered to be local.

4.2.5 International status with country highlights

International

OSPAR Convention:	Not Listed
EC Habitats Directive:	Annex V (2013) – unfavourable. Past and current threats (Habitats Directive Article 17 codes): Climate change (M01.01, M01.04), Construction (D03.03, J02.02.01), Contaminant pollution (H01), Eutrophication (H01.05), Fishing (F02).
CITES:	Not listed
Bern Convention:	Appendix III (2002)
Bonn Convention:	Not listed

IUCN Criteria: A2bcd

Global IUCN Red List Category: LC

European IUCN Red List Category: LC

HELCOM Red List: CR

National IUCN Red List classifications

Denmark: Totally protected since 4.2.2001 / VU

Estonia: Protected by the law (second category) / VU

Finland: Totally protected by national legislation / CR

Germany: – / 2 (Endangered, freshwater)

Latvia: Commercial fishing and angling rules (closed season, minimal landing size), regulation nr. 45 and 396 / –

Lithuania: Fishing is not allowed from 1 March until 15 May. Size limit for fishing is 29 cm total length / –

Poland: The species does not exist in polish coastal waters; inland population of the species is supported by stocking; minimum landing size 30 cm; fishing is not allowed from 1 March until 31 May / DD

Russian Federation: Fisheries regulations / –

Sweden: Minimum landing size 30 cm in Åland Sea / LC

4.2.6 Distribution summary

Coastal grayling is distributed in the Baltic Sea, from the Bothnian Sea to the Bothnian Bay, in rivers and coastal areas. However, the complete range of grayling covers nearly entire Europe (Figure 4.2).

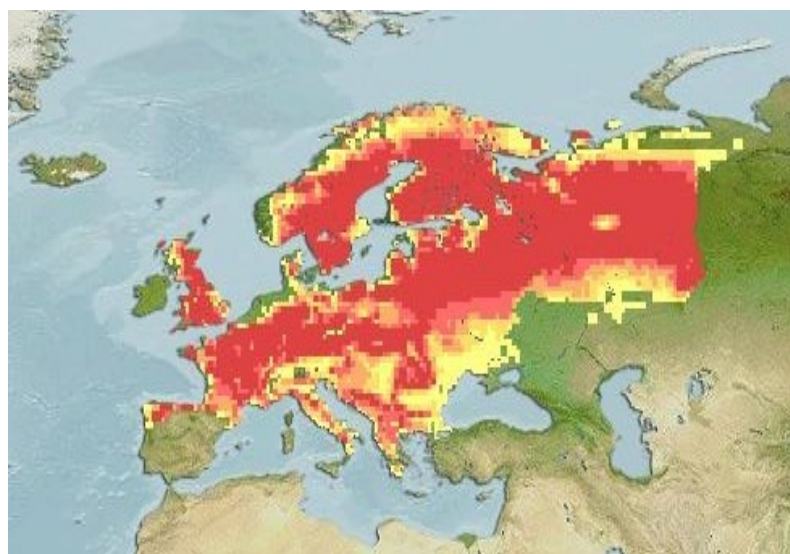


Figure 4.2. Distribution of grayling (Froese and Pauly, Fishbase 2015).

4.2.7 Country-by-country changes in distribution

The historical distribution of coastal grayling covers the area from the southern Bothnian Sea to the northernmost reach of the Bothnian Bay. Currently, the distribution, is mainly limited to the Bothnian Bay, and occurrence of grayling in both fishery catches and monitoring is sporadic at best. Registered catches in the Swedish commercial fishery are currently < 10 kg annually, compared to 1918 kg reported in 1930 and around 1000 kg in 1968–1969. In general, the stocks have decreased substantially, especially in the Finnish coastal area, where the known southern populations in the Bothnian Sea have become nearly extinct (Urho *et al.*, 2010). In Sweden, coastal grayling still occur in many areas, although stocks are considered to be weak.

Coastal migratory grayling are declining in several Swedish rivers in the northern Bothnian Sea and Bothnian Bay (Nordwall and Carlstein, 2001, unpub.). The decrease in the Baltic Sea area stocks is believed to have already started in the 1950s. Enquiries made to coastal fishers in Sweden (Jensen and Alanära, 2006) and Finland (Ruotsalainen, 2011) show that the abundance of grayling has decreased during the last 20 years in Sweden, and more markedly and for a longer period in Finland. The exact level of the decrease is difficult to estimate as a result of the low catchability of the species in monitoring, and the current low population density. However, the range of decrease is roughly estimated at 50–90%. The situation for sea-spawning grayling is much worse than that for anadromous grayling, although, over the last 5-year period, stocks in certain areas of the Swedish coast of the Bothnian Bay are possibly recovering (County Administration Board of Norrbotten, unpub.). Anadromous grayling probably still occur in a few rivers in Finland and Sweden. However, the occurrence of non-anadromous grayling in rivers makes it difficult to attain precise abundance estimates.

4.2.8 Threats

The coastal grayling is considered to be CR in the Baltic Sea area (HELCOM, 2013a). The exact reasons for stock decreases of coastal grayling are not well known. Building of dams in rivers for hydropower, increased fishing pressure, decreased water quality in rivers and coastal areas, and increased predation by seals and cormorants, have all been suggested as potential causes of the decline. Some of the potential threats are also connected to large-scale changes in rivers and coastal areas, particularly related to climate change and eutrophication.

4.2.9 Conclusion – future

Management recommendations: Due to small number of individuals, a fishery is not recommended. As the main catches of grayling today take place as occasional bycatch in other fishery that will not be banned in the near future, there should also be recommendations for how these fishery could be better regulated from a grayling perspective.

The following fishing restrictions on critical areas are suggested to protect the endangered stocks:

- No fishing with gillnets or traps in recognized grayling habitats all year-round.
- No fishing with gillnets or traps adjacent to grayling river mouths during spawning migration.
- Zoning approach to recreational fishery: in areas with weak stocks, all fish should be released; in areas with moderately weak-strong stocks, a bag limit and size window, or minimum legal size of 40 cm.

- Collaboration with commercial and subsistence fishers in the most essential coastal grayling areas to develop measures to drastically reduce bycatches of grayling.

Since 2016, the coastal grayling has been protected by law in Finland. Introductions and preservation of stock-specific genes in cultivation are recommended before the remaining stocks are lost. Protection of spawning and larval habitats, as well as restoration and remediation measures should also be considered. Large-scale actions, such as reduction of eutrophication in rivers, removal of migration barriers, land-use changes in drainage areas to minimize acidification in rivers and estuaries, and reduction in sediment load, would most likely benefit the species. An increase in scientific knowledge of the species would be needed to measure the effect of such actions as a management measure.

4.3 Ide

4.3.1 Taxonomy

Class: Actinopterygii
Order: Cypriniformes
Family: Cyprinidae
Scientific name: *Leuciscus idus*

Common names

DE: Aland; DA: Rimte; EE: Säinas; FI: Säyne; LV: Ālants; LT: Meknė; PL: Jaź; RU: Язь; SV: Id.

General characteristics

Maximum total length: 85 cm
Maximum weight: 2.2 kg
Maximum age: 29 years
Generation time: 7-10 years

4.3.2 Life cycle and migrations

Ide are rheophilic cyprinids that occur naturally in the large lowland rivers and nutrient-rich lakes of Europe and Asia (Järvalt *et al.*, 2003). Large numbers of ide can also be found in the brackish coastal waters of the Baltic and Caspian seas (Järvalt *et al.*, 2003; Bogutskaya and Naseka, 2006). A population living near the Öresund Strait in the Baltic Sea is probably an example of ide living at the upper limit of their salinity tolerance (i.e. 10–20 ppt; Cala, 1970). Ide are known to spawn in rivers, lakes, estuaries, bays, and, reportedly, even in the open coastal areas of the Baltic Sea (Erm *et al.*, 1970; Müller and Berg, 1982; Järvalt *et al.*, 2003). It can be classified as facultatively anadromous. Anadromous migrations to rivers and lakes may start under ice, with spawning beginning some days after ice breakup, at water temperatures of 4–6°C (usually in the middle of April; Järvalt *et al.*, 2003). Spawning in flooded shallow areas lasts for only 3–7 d, after which adults migrate back to the sea (Järvalt *et al.*, 2003). Ide larvae hatch at the beginning of May. Progeny of anadromous individuals stay in freshwater from a few weeks to a few months, or up to a year (depending on the salinity of the sea) before descending to the sea (Cala, 1970; Rohtla *et al.*, 2015). Migrations at sea can be relatively extensive, but homing seems to be profound (Johnson, 1982). The growth rate of ide is one of the highest among cyprinid fish, making ide a desirable target for commercial and, especially, recreational fishery in some countries (Järvalt *et al.*, 2003; Anon., 2008a; Krejszeff *et al.*, 2009).

4.3.3 Stock structure and population dynamics

No population estimates are available for coastal ide populations in the Baltic Sea. Populations are most likely declining or stable, depending on the region.

Estonia

Catches of coastal ide peaked in the 1980s at 177 t per year, but ide abundance and catches have constantly decreased since then (Vetemaa *et al.*, 2006, 2010; Eschbaum *et al.*, 2016). It is likely that overfishing led to the collapse of most ide spawning stocks in Estonia (Vetemaa *et al.*, 2001). An increase in juvenile and adult ide has been recorded

in some coastal areas in recent years, but the historically abundant Matsalu and Saunja Bay spawning stocks have not recovered (Eschbaum *et al.*, 2016). In general, it seems that large variations occur in reproductive success and recruitment between years.

Finland

The ide was previously a popular species for household use, but it has fallen into dis-favour, concurrent with the general decrease in the appreciation of cyprinids for human consumption. Some ide are still caught for the market in the Archipelago Sea, the Gulf of Finland, and in estuaries of the northern Gulf of Bothnia. Ide stocks have been declining locally owing to eutrophication, dam building, and water-level regulation. Some populations have even vanished as a result of acidification (Anon., 2008a).

Latvia

Catches of ide have decreased in the traditional fishing areas in the coastal waters of the Gulf of Riga (J. Birzaks, pers. comm.).

Poland

In coastal waters, ide is present in rather small numbers. Additionally, because of its visual similarity to roach (*Rutilus rutilus*; more recognizable species), it might not be distinguished from roach and/or sold as roach in local markets. This makes any monitoring supported by fishery data impossible.

Sweden

Ide is rare in different monitoring areas, but there does not seem to be an overall decline since 2001 in any area. In the commercial coastal fishery, catches are very low and with no trend since 1999 (J. Olsson, pers. comm.).

4.3.4 Genetics

Not much is known about the population genetic structure of anadromous ide. The only published study to date shows that in Denmark ide exhibits the genetic population structure of both a typical freshwater and a typical anadromous species (Skovrind *et al.*, 2016). The study also found low levels of genetic differentiation among anadromous populations from Denmark and Sweden, which could suggest that ide not only migrate along the coastline, but can cross deeper sea areas as well. As some degree of population genetic structure was evident, it suggests precise homing to natal streams.

4.3.5 International status with country highlights

International

OSPAR Convention:	Not listed
EC Habitats Directive:	Not listed
CITES:	Not listed
Bern Convention:	Not listed
Bonn Convention:	Not listed
IUCN Criteria:	Not listed
Global IUCN Red List Category:	LC
European IUCN Red List Category:	LC

HELCOM Red List: Not listed

National IUCN Red List classifications

Denmark:	– / LC
Estonia:	Minimum legal size in the sea and directly connected rivers (38 cm TL) / LC.
Finland:	– / LC
Germany:	In coastal waters, no minimum legal size, but in adjacent fresh-water 25 cm TL / not listed.
Latvia:	Minimum legal size (30 cm TL) / not listed
Lithuania:	Minimum legal size (30 cm for recreational and 45 cm for commercial fishers in the Curonian lagoon) / not listed
Poland:	Minimal landing size (25 cm, commercial and recreational fishery) / not listed
Russian Federation:	Data not available
Sweden:	– / Not listed



Figure 4.3. Distribution of ide (Freyhof and Kottelat, 2008b). Diadromous populations are observed mainly in the Baltic Sea area where relatively low salinity in the coastal area es the anadromous life cycle.

4.3.6 Distribution summary

Facultatively anadromous populations (Figure 4.3, entire distribution): Denmark, Estonia, Finland, Germany, Latvia, Lithuania Poland, the Russian Federation, Sweden, Caspian Sea countries (Table 4.3).

Table 4.3. Distribution of anadromous ide in the Baltic Sea by country.

Country	Distribution
Denmark	Common in coastal brackish waters.
Estonia	Common in the Väinameri Sea region and Saaremaa and Hiiumaa islands. The most abundant runs of anadromous ide occur in Käina and Vaemla bays, and in the Kõrgessaare region (all in Hiiumaa Island). Historically abundant stocks that spawned in Matsalu and Saunja bays (and in their rivers) are currently very low in number. Rare in coastal waters of northern and southwestern Estonia.
Finland	Occurs in all coastal brackish waters and several inland waters.
Germany	Common in almost all brackish areas with coastal streams and rivers (H. Winkler, pers. comm.).
Latvia	More common in the west coast of the Gulf of the Riga. Historically found in ca. 76 rivers, but currently only in about 40 rivers (Birzaks <i>et al.</i> , 2011). Anadromous ide spawns in lakes and rivers (e.g. Lake Engure).
Lithuania	Data not available.
Poland	Present in small number in Vistula and Szczecin Lagoon; exists in the coastal-zone of the Baltic Sea in the vicinity of the mouths of rivers flowing by coastal lakes located along the Polish coast.
Russia	Data not available.
Sweden	Found throughout the coastline from the Bothnian Bay to southern Sweden, mainly in the brackish parts and in moving water (sounds, narrow passages). Often found migrating up rivers and tributaries for spawning in spring (J. Olsson, pers. comm.).

4.3.7 Country-by-country changes in distribution

The following trends have been observed country-by-country:

Denmark:	Not ranked/unknown
Estonia:	Declining/unstable
Finland:	Declining
Germany:	Stable
Latvia:	Common species in the coastal waters, but populations are low in number; the number of rivers inhabited by ide has declined
Lithuania:	Data not available
Poland:	Unknown
Russian Federation:	Data not available
Sweden	Stable/unknown

4.3.8 Threats

The problems associated with ide living in the Baltic Sea generally concern: (i) dysfunctional freshwater recruitment, as many spawning areas have been acidified (Eriksson and Müller, 1982; Anon., 2008a), or (ii) anthropogenic or natural habitat change (e.g. damming, ground uplift, and dense vegetation at river mouths). Eutrophication continues to deteriorate the brackish-water spawning areas (e.g. inner bays and estuaries).

Water level regulation in rivers and overfishing negatively influence ide in some regions. Climate change will decrease the extent and duration of spring floods which will negatively affect ide spawning in flooded areas. Ide seems to be more sensitive to environmental perturbations than the closely related roach.

Ide monitoring programmes in the Baltic Sea countries are shown in Table 4.4.

Table 4.4. Monitoring programmes and data for anadromous ide in the Baltic Sea countries.

Country	Monitoring and data
Denmark	Data not available.
Estonia	Monitored annually in several sea areas as a part of the national coastal sea fish monitoring programme; no data from freshwater; official landings from the sea and rivers.
Finland	Caught in coastal gillnet monitoring programmes; official landings from the sea and rivers.
Germany	There are no monitoring and catch statistics for ide; some data are available from other monitoring programmes running in brackish and freshwater areas.
Latvia	Monitored in coastal and inland waters; official landings from the sea and rivers.
Lithuania	Data not available.
Poland	Piecemeal data achieved from Vistula and Szczecin Lagoon as a part of monitoring conducted within multiannual programme for collection of fishery data.
Russian Federation	Data not available.
Sweden	Caught in coastal gillnet monitoring programmes, but at low densities. Population estimates hence uncertain.

4.3.9 Conclusion – future

Management recommendations:

- Monitoring programmes should be initiated for unmonitored stocks.
- To maintain/increase abundance in stable-state stocks, or stop the decline in declining stocks, fishing regulations should be implemented.
- Totally closed spawning seasons and a minimum landing size should be established.
- To enhance natural reproduction of anadromous ide, channelized rivers should be restored to naturally meandering rivers.
- Levels of anthropogenic nutrient pollution should be decreased to facilitate the recovery of anadromous and brackish-water spawning populations.
- Restocking is recommended, but only the local broodstock should be used.
- Since the reasons for declining stocks are not well known, it is necessary to increase the knowledge of ide life history and ecology, in order to suggest meaningful action plans.

4.4 Perch

4.4.1 Taxonomy

Class: Actinopterygii
 Order: Perciformes
 Family: Perciidae
 Scientific name: *Perca fluviatilis*

Common names

DE: Barsch; DA: Aborre; EE: Ahven; FI: Ahven; LV: Asaris; LT: Ešerys; PL: Okoń;
 RU: Окунь; SV: Abborre.

General characteristics

Maximum standard length: 60 cm
 Maximum weight: 4.8 kg
 Maximum age: 22 years
 Usual generation time: 6-10 years
 IUCN Criteria: LC



Figure 4.4. Distribution of perch (Freyhof and Kottelat, 2008c). Diadromous populations are observed mainly in the Baltic Sea area where the relatively low salinity in the coastal area es an anadromous life cycle.

4.4.2 General overview

Perch are freshwater fish that also inhabit the brackish Baltic Sea. Perch is an important target species for commercial fisheries in coastal waters. Populations inhabiting the Baltic Sea generally spawn in the sea (Pihu *et al.*, 2003; Wastie, 2014), but anadromous migrations occur in some regions. In Sweden, anadromous perch have been recorded with trapnets in River Ängerån (Johnson, 1982) and River Andersbäcken (Berg, 1982). These rivers act as migratory corridors that are used to reach the lakes where spawning occurs (0.5–8 km from the sea). Adult perch descend to the sea soon after spawning,

and young-of-year follow in August–October (Eriksson and Müller, 1982). In addition, Wastie (2014) sampled perch along the Swedish Baltic coast and demonstrated that 25% were freshwater recruits; although only a few sites contained freshwater recruits and the majority contained brackish-water recruits only.

In Germany, perch is, to a large extent, anadromous (migrations are documented by tagging experiments; H. Winkler, pers. comm.). Furthermore, perch is also known to migrate between brackish and freshwater in Lithuania and Latvia (Ložys, 2013). However, it remains to be determined whether these perch are really anadromous or just interhabitat shifters. In Estonia, documented occurrences of anadromy are lacking, but according to anecdotal evidence, coastal perch ascend to Harku and Käsnu lakes for spawning. Perch spawning migrations to semi-enclosed Käina, Vaemla, and Mullutu bays are also well known. These bays are flooded with freshwater in spring, but otherwise salinities of 1–3 ppt occur. In Denmark, some perch are also thought to be anadromous, but details about their spawning behaviour are unknown (Skovrind *et al.*, 2013). In the Polish coastal area perch occur mainly in Vistula and Szczecin Lagoons and in the nearshore zone up to 300 m from the coastline (ICES 2003). Perch populations in most of the Baltic Sea coastal areas are stable or increasing (HELCOM, 2015).

4.5 Pike

4.5.1 Taxonomy

Class: Actinopterygii
 Order: Esociformes
 Family: Esocidae
 Scientific name: *Esox lucius*

Common names

DE: Hecht; DA: Gedde; EE Haug; FI: Hauki; LV: Īidaka; LT: Iydeka; PL: Szczupak;
 RU: Щыка; SV: Gädda.

General characteristics

Maximum standard length: Ca. 100 cm
 Maximum weight: Ca. 25–28 kg
 Maximum age: 30 years
 Usual generation time: 6–10 years

Northern pike is widely distributed across the Holarctic, both in inland waters and in coastal areas of several brackish seas (Craig, 1996; Bogutskaya and Naseka, 2006; Larsson *et al.*, 2015). In the brackish Baltic Sea, pike displays a facultatively anadromous life history, i.e. populations or individuals within populations can display anadromous behaviour (Müller, 1986; Engstedt *et al.*, 2010; Rohtla *et al.*, 2012). Pike is a recreationally important species throughout its range. Commercial importance is limited to the brackish seas and larger inland lakes and rivers throughout the range (Raatt, 1988; Craig, 1996; Pihu and Turovski, 2003).

4.5.2 Life cycle and migrations

Pike displays three general life-history types: freshwater resident, anadromous, and brackish-water resident. It can be classified as facultatively anadromous. Adults can tolerate salinities up to 15 ppt, but spawning mostly occurs in freshwater or in salinities up to 8.5 ppt (Raatt, 1988; Jørgensen *et al.*, 2010). Spawning migrations of anadromous pike generally do not exceed 15 km in freshwater (Müller, 1986; M. Rohtla, unpub. data), but long distance migrations in brackish water are possible and can reach ca. 60 km (Karås and Lehtonen, 1993; Bekkevold *et al.*, 2014; M. Rohtla, unpub. data). The highly migratory capacity of some individual pike has been verified by reported examples of 40–140 km migrations (Burkholder and Bernard, 1994; Koed *et al.*, 2006).

Pike is a top predator in aquatic foodwebs that mostly consumes fish, but can prey on different types of food items (from insects, to birds and small mammals; Craig, 1996). Pike spawn in spring (March–May) when the daily average water temperatures reach 4–13°C (Pihu and Turovski, 2003; Nilsson *et al.*, 2014). Anadromous individuals make their way to rivers, creeks, ditches, and lakes (or their floodplains) where they spawn on dead vegetation in shallow water (Nilsson *et al.*, 2014). Anadromous pike can mix with resident pike in freshwater spawning grounds (Müller, 1986). Progeny of anadromous individuals spend from a few weeks to several years in freshwater nursery grounds, then descend to the sea (Johnson and Müller, 1978; Engstedt *et al.*, 2010; Rohtla *et al.*, 2012; M. Rohtla, unpub. data). Maturity is usually reached at an age of two to five years (Pihu, 1972).

4.5.3 Stock structure and population dynamics

No population estimates are available for anadromous pike inhabiting the brackish Baltic Sea. This is partly owing to the fact that anadromous pike mix with brackish-water resident populations in the sea. However, in general, pike abundance is currently very low compared to historical levels. This is reflected in official catch statistics and national surveys from Denmark, Estonia, Finland, and Sweden (Pihu and Turovski, 2003; Nilsson *et al.*, 2004; Jacobsen *et al.*, 2008; Lehtonen *et al.*, 2009; Ljunggren *et al.*, 2010). The reasons behind these declines are complex. In freshwater, the main negative factors affecting anadromous pike spawning and recruitment are probably destroyed wetlands, dammed and channelized rivers, anthropogenic pollution, and poaching. In brackish water the main negative factors probably are anthropogenic eutrophication and overfishing. Overfishing is also a negative factor for anadromous pike in the sea. It has been demonstrated that eutrophication has deteriorated brackish-water spawning areas (Eriksson *et al.*, 2009) and positively influenced the abundance of the three-spined stickleback (*Gasterosteus aculeatus*), which is known to prey on pike eggs and larvae (Nilsson, 2006).

In the Väinameri Sea, which is the most productive pike region in the Estonian coastal sea, commercial catches have declined from a mean of ca. 70 t per year in 1970–1980s, to 20 t in the 1990s, and 10 t in the 2000s. Historically, the brackish-water spawning areas of pike were widespread in the Väinameri Sea (Erm *et al.*, 1970). However, currently, it seems that brackish-water-spawning pike have a very low abundance, and only some small spawning stocks exist. Only 10% of the adult pike sampled in the Väinameri Sea were born in brackish water, whereas the other 90% were of freshwater origin (Rohtla, 2015). In comparison, Engstedt *et al.* (2010) reported that 46 and 54% of the pike sampled in the Swedish coastal areas were of freshwater and brackish-water origin, respectively. This indicates that the extent of anadromy may vary significantly between regions (and between years; J. Olsson, pers. comm.), which may reflect the quality and/or the availability of fresh- and brackish-water spawning areas. A high pike abundance is, therefore, expected in regions where spawning areas from both of these biomes are readily available for the fish.

4.5.4 Genetics

The genotypic distribution of living Baltic Sea pike is incompatible with that of a single panmictic population. Isolation by distance shapes the genetic structure (Laikre *et al.*, 2005; Wennerström *et al.*, 2013). A genetic patch size of 100–150 km could be used as a preliminary basis for identifying management units for Baltic Sea pike (Laikre *et al.*, 2005). However, anadromous individuals originating in closely situated streams can display genetic differences that most likely result from precise homing mechanisms (Larsson *et al.*, 2015). Genetic risks include the loss of local adaptation, and genetic swamping as a consequence of stocking with non-local individuals.

4.5.5 International status with country highlights

International

OSPAR Convention:	Not listed
EC Habitats Directive:	Not listed
CITES:	Not listed
Bern Convention:	Not listed

Bonn Convention:	Not listed
IUCN Criteria:	Not listed
Global IUCN Red List Category:	LC
European IUCN Red List Category:	LC
HELCOM Red List:	Not listed

National IUCN Red List classifications (HELCOM Red List)

Denmark:	Stocking of specimens / LC
Estonia:	Protected from fishing 1 March–30 April (in the sea) and 15 March–10 May (in freshwater, except in larger lakes where different dates apply); minimum landing size (45 cm TL) / LC
Finland:	Stocking of specimens / LC
Germany:	Protected from fishing 1 March–30 April; minimum landing size (50 cm TL) / LC
Latvia:	Protected from fishing 1 March–30 April; minimum landing size (50 cm TL); regular restocking / –
Lithuania:	– / LC
Poland:	Protected from fishing 1 January/1 March–30 April/15 May (depends on the area), minimum landing size (45/50 cm, depends on the area) / not listed
Russian Federation:	Data not available
Sweden:	Bag limit and minimum (40 cm) and maximum (75 cm) size limits for recreational fishers; protected from fishing 1 April–31 May / –



Figure 4.5. Distribution of northern pike (NatureServe, 2013c). Diadromous populations are observed mainly in the Baltic Sea area (and perhaps also in the Caspian Sea), where relatively low salinity in the coastal area es an anadromous life cycle.

4.5.6 Distribution summary

Facultatively anadromous populations (Figure 4.5): Denmark, Estonia, Finland, Germany, Latvia (rare); Lithuania (rare), Poland, the Russian Federation, Sweden, and Caspian Sea countries (Table 4.5).

Table 4.5. Distribution of anadromous pike in the Baltic Sea by country.

Country	Distribution
Denmark	Common in coastal waters.
Estonia	Very common in the Väinameri Sea region and Saaremaa and Hiiumaa islands. Most abundant runs of anadromous pike occur in rivers/streams that empty into the Matsalu and Saunja bays (e.g. Kasari, Rannamõisa, Tuudi, Penijõgi, Liivi, Taebala, Salajõgi rivers). Larger runs probably also occur in Paadrema and Võlupe rivers, and in the Laidevahe Bay system. Rare in coastal waters of northern and southwestern Estonia.
Finland	Common in coastal waters.
Germany	Very common in all streams and rivers running into the Baltic Sea; rare in inner coastal waters in the western part as a result of the high salinity in adjacent Baltic areas. Common or abundant in the lagoons and estuaries on the east coast of the German Baltic Sea. The highest densities are found in lagoons around Rügen Island (Helmut Winkler, pers. comm.).
Latvia	Not very common in coastal waters, distributed in all types of inland waters (J. Birzaks, pers. comm.).
Lithuania	Common in the Curonian Lagoon, but not very abundant. Mostly inhabits shallow waters along Nemunas River delta. spawning occurs in the tributaries of Nemunas River delta. Common fish in Sventoji River, which flows directly to the sea in the northern part of the Lithuanian coastline. Rarely inhabits brackish coastal waters (L. Lozys, pers. comm.).
Poland	Low numbers in Vistula Lagoon; common in Szczecin Lagoon and in the Bay of Puck.
Russia	Data not available.
Sweden	Common along the whole Swedish coast down to Blekinge (J. Olsson, pers. comm.).

4.5.7 Country-by-country changes in distribution

Pike abundance in the Baltic Sea is currently very low compared to historical levels. This is reflected in official catch statistics and national surveys from Denmark, Estonia, Finland, Germany, and Sweden (Winkler, 2002; Pihu and Turovski, 2003; Nilsson *et al.*, 2004; Jacobsen *et al.*, 2008; Lehtonen *et al.*, 2009; Ljunggren *et al.*, 2010; Eschbaum *et al.*, 2016).

Estonia

In the 1930s, > 400 t of pike were caught annually from coastal waters (Pihu and Turovski, 2003). During the past two decades, < 120 t were caught annually, with lows at 10–20 t per year. In recent years, localized occasional increases in pike abundance have been documented on account of favourable spring-spawning conditions for anadromous individuals (Eschbaum *et al.*, 2016). However, once an increase in numbers is evident, it will soon start to decrease as a result of increased fishing pressure from commercial and recreational fishers.

Finland

Pike stocks have diminished, but, in general, pike is regarded as a LC species, and most populations are self-sustaining.

Germany

At the end of the 1960s, pike landings from the German commercial fishery reached a peak of about 300 t. Since then, landings have shown a generally decreasing trend, reaching the lowest value of 45 t in 2010. However, in recent years, landings have increased, and up to 100 t of pike are now landed annually.

Lithuania

Commercial landings during last two decades have been stable, and in the range of 10 t per year in the Lithuanian part of the Curonian Lagoon and 22 t for the entire lagoon (including the Russian part). However, during the 1970s–1980s, landings were in the range of 35 t on average for the Lithuanian part and 66 t for the whole lagoon. Pike in the lagoon are regarded as self-sustaining and a LC species.

Poland

In the past, the anadromous population of pike living in the Bay of Puck could spawn in spring-flooded meadows. However, as those areas were lost, natural populations became almost extinct (Skóra, 1993), and they are currently supported by stocking.

Sweden

Overall patterns are unclear. Stocks appear to be in decline in more exposed archipelago areas, whereas they seem to be more stable in the Gulf of Bothnia and Baltic Proper inner and middle waters. However, in the middle and northern parts of the Baltic Proper, there seems to be a decline in recent years (J. Olsson, pers. comm.).

4.5.8 Threats

Pike can spawn in fresh- and brackish water if the required adaptations are present in the given population. This is undoubtedly beneficial in coping with environmental perturbations. However, because virtually nothing is known about pike's ability to switch between these spawning biomes, we must take the conservative, and most plausible view that anadromous and brackish-water-spawning populations are reproductively isolated. This means that freshwater and brackish-water spawning areas are equally important in maintaining viable pike stocks in the Baltic Sea.

Pike inhabiting the Baltic Sea are threatened by numerous anthropogenic and natural factors. Climate change will most likely have a negative effect on anadromous populations as springs with extensive and prolonged floods will become rarer. This coupled with the fact that many rivers and streams are completely or partly channelized will substantially decrease available spawning areas for pike. In freshwater, dams and other obstacles on rivers continue to obstruct spawning migrations. Anthropogenic pollution is a more serious threat in freshwater habitats. Nutrients tend to accumulate in river mouths and promote high plant growth and sedimentation rates, which can result in natural migration barriers for migrating fish (especially in smaller rivers). In addition, poaching during spring-spawning continues to be a large problem (e.g. in Estonia).

In brackish waters, anthropogenic eutrophication continues to deteriorate spawning areas and promote the increase in some regions of mesopredators, such as sticklebacks.

Finally, overfishing is still a problem in some regions, as it does not allow pike stocks to build up after successful spawning seasons. Pike populations are not well monitored in coastal areas (Table 4.6).

Table 4.6. Monitoring programmes and data for anadromous pike in the Baltic Sea countries.

Country	Monitoring and data
Denmark	Data not available.
Estonia	Monitored annually in several sea areas as a part of the national coastal sea fish monitoring programme; only occasional data from electrofishing surveys in rivers/wetlands; official landings from the sea.
Finland	Monitored as a part of general coastal fish monitoring programme.
Germany	No special monitoring for pike, but the reproduction of pike is studied in some coastal areas; electrofishing surveys in the most important streams and rivers (EU-water frame directive).
Latvia	Not very common in coastal waters, official landings from the coastal and inland waters; angling data from a few lakes and rivers.
Lithuania	There are official landings available from the Curonian Lagoon (for commercial fishery). Monitoring programmes barely cover pike stock assessments, because they are not conducted in shallow bays, along coastline in shallow waters or in the Nemunas River tributaries.
Poland	Official landings from the sea.
Russian Federation	Data not available.
Sweden	Information both from coastal gillnet monitoring and coastal commercial fishery.

4.5.9 Conclusion – future

Management recommendations:

- Better data are needed to assess the status of stocks.
- To maintain sustainable pike fishery and stocks, fishing regulations, stocking, and habitat restoration should be used or continued.
- Totally closed spawning season and minimum/maximum landing size should be established.
- Poaching should be decreased.
- Catch-and-release angling should be promoted in regions where it is not yet popular.
- Pollution from pulp mills and mining activities should be minimized.
- Levels of anthropogenic nutrient pollution should be decreased to facilitate the recovery of anadromous and brackish-water spawning populations.
- To enhance the natural reproduction of anadromous pike, channelized rivers should be restored to naturally meandering rivers.
- Pike hatcheries (*sensu* Nilsson *et al.*, 2014) could be constructed in suitable rivers and streams to increase the recruits from any given spawning area. It has been suggested that hatcheries would also decrease the amount of nutrients reaching the sea.
- With the brackish-water spawning stocks, the main reason for decreasing stocks are eutrophication and overfishing. To protect the remaining stocks, local fishing regulations are suggested.
- Restocking is recommended, but only the local broodstock should be used.

4.6 Roach

4.6.1 Taxonomy

Class: Actinopterygii
Order: Cypriniformes
Family: Cyprinidae
Scientific name: *Rutilus rutilus*

Common names

DE: Plötze; DA: Skalle; EE: Särg; FI: Särki; LV: Rauda; LT: Kuoja; PL: Płoć; RU: Плотва; SV: Mört.

General characteristics

Maximum standard length: 50 cm
Maximum weight: 1.8 kg
Maximum age: 14 years
Usual generation time: 6-10 years
IUCN Criteria LC

4.6.2 General overview

Roach are freshwater fish that also inhabit the brackish Baltic Sea (Figure 4.6). Populations inhabiting the Baltic Sea generally spawn in the sea or in estuaries and rivers/lakes in April–May (Vetemaa *et al.*, 2003; Härma *et al.*, 2008). However, little is known about their anadromous migrations. Anadromous spawning migrations of roach are reported from Estonia, Finland, Latvia and Sweden (Berg, 1982; Johnson, 1982; Vetemaa *et al.*, 2003; Anon., 2008b; J. Birzaks, pers. comm.), where hundreds or even thousands of individuals can ascend even one small stream. This spring mass migration of roach is targeted by many recreational fishers, who mostly dry the fish for later consumption (e.g. in Estonia and Latvia). Adult roach descend to the sea soon after spawning, and young-of-year follow in August–October (Johnson, 1982). In Germany, roach is one of the most abundant freshwater species in brackish waters. There are resident and (semi-)anadromous forms. Migrations between spawning places in oligohaline areas (salinity < 3 ppt) and the adjacent Baltic Sea areas have been well documented through tagging experiments.



Figure 4.6. Distribution of roach (Freyhof and Kottelat, 2008d). Diadromous populations are observed mainly in the Baltic Sea area (and perhaps also in the Caspian Sea), where relatively low salinity in the coastal area es an anadromous life cycle.

4.7 Smelt

4.7.1 Taxonomy

Taxonomical group [after Belyanina (1969)]

Class: Teleostomi, subclass Actinopterygii
Order: Clupeiformes, suborder Salmonoidei
Family: Osmeridae, subfamily Osmerinae
Scientific name: *Osmerus eperlanus*

Taxonomical group [after Wikipedia (2016)]

Class: Actinopterygii
Order: Osmeriformes
Family: Osmeridae
Scientific name: *Osmerus eperlanus*

Taxonomic notes

Osmerus eperlanomarinus

Salmo eperlanomarinus Bloch, 1782, bracknors Slom

Salmo eperlanus Linnaeus, 1758.

Common names

EN: Smelt; FI: Kuore; SV: Nors; DE: Stint; ET: Meritint; PL: Stynka

General characteristics

Smelts are a family of small fish, Osmeridae, found in the Atlantic and Pacific oceans. *Osmerus mordax* occupies North American waters, in both the Atlantic and Pacific oceans, with its Pacific range extending southward to Japan (Belyanina, 1969). *Osmerus eperlanus* is found from the White Sea in the west, to the Atlantic, and southward to the Bay of Biscay.

Belyanina (1969) summarized existing taxonomic information. Three forms of the boreal smelt were generally recognized: *O. eperlanus* (North Europe and Baltic), *O. dentax* of the North Pacific, Arctic, and into the White Sea, and *O. mordax* of the western Atlantic. Further investigations first reduced “*dentax*” to a subspecies of “*mordax*”, and later determined that the genus *Osmerus* included only one species: *O. eperlanus* (L.) (Bigelow *et al.*, 1963 and Mc Allister, 1963, as reported in Belyanina, 1969).

Froese and Pauly (2016) also refer to two former nominal subspecies: *Osmerus eperlanus* and *Osmerus eperlanus schonfoldi* (Rutty, 1772). The former encompasses primarily lacustrine smelt that are also recorded from the coasts and drainage of the White and Barents seas, and westward, through the Baltic Sea to Denmark. The latter encompasses sympatric smelt in parts of Poland, Denmark, and the Baltic, which are primarily anadromous.

4.7.2 Life cycle and migrations

Smelt (*Osmerus eperlanus* L.) live in coastal and estuarine waters of Europe, from the Baltic Sea to the Bay of Biscay (Figure 4.7). Its range extends from the White Sea in the north to the west coast of France in the south, and includes the Baltic Sea, southern North Sea, and the British Isles. The Gironde Estuary used to be the southern limit of its distribution, but that population has been extirpated (Pronier and Rochard, 1998). Landlocked populations occur in lakes of coastal areas in the North, Baltic, White, and Barents seas, and to about 68°N in Scandinavia.

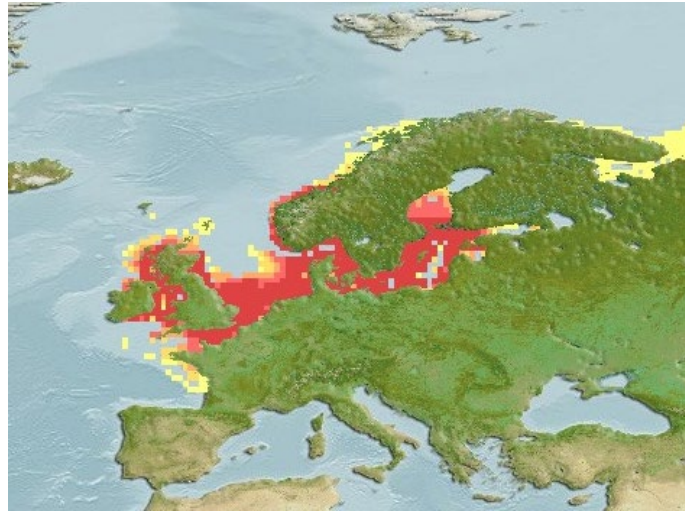


Figure 4.7. Distribution range of smelt in Fishbase (Kottelat and Freyhof 2007).

Arguably, there are two forms of smelt: one that spawns in freshwater (anadromous) and one that spawns in brackish waters of 1–4 psu (Špilev and Turovski, 2003), which can be regarded as semi-anadromous (*sensu* Elliot *et al.*, 2007). Adult fish migrate into the upper reaches of estuaries in spring to spawn. Smelt are relatively weak swimmers, and, therefore, access from estuaries into freshwater for spawning can be disrupted by weirs and other structures. Even major riffles can be barriers to migration. Therefore, bypass devices designed to assist other diadromous fish in overcoming physical barriers are unlikely to be effective for smelt. Smelt congregate in the underflows of stronger currents in order to spawn over areas of sand. Spawning takes place from the end of February to March, with the timing related to latitude and water temperature. Water temperatures of 6°C are common during spawning time in Irish locations. Smelt can lay as many as 40 000 eggs. After spawning, there are often mass mortalities. Smelt feed mainly on small plankton, crabs, benthic invertebrates, and even its own young.

Adult smelt can grow up to 30 cm in length and are slender and slightly flattened on either side. Smelt have a slightly translucent body. The back and sides are grey-green to pink in colour, and the flanks bright silver. The tail has a dark border. Smelt live for up to six years.

Smelt can migrate through coastal waters. This can be seen through the impingement of smelt on power station cooling-water screens located in full-strength seawater, at least 5 km from the nearest lower salinity area, and 25 km from the nearest known smelt population (Colclough and Coates, 2013). This means that they have the ability to recolonize estuaries, and that estuary/river complexes may not support discrete stocks. However, areas exist where smelt are relatively abundant in some rivers but are apparently absent from other nearby rivers. This includes situations where smelt are found in adjoining waters on either side of a waterbody where smelt are absent.

4.7.3 Stock structure and population dynamics

No population estimates are available for coastal smelt populations. No formal assessment is made of smelt populations in UK, although Maitland (1979) and Maitland and Lyle (1991) have described the species as threatened because it has disappeared from many rivers during the last two centuries. In many countries, the smelt stock assessment situation is similar to this. Maitland (2003) reviewed information on smelt in England (UK) and concluded that thriving populations exist in several areas. It is uncertain whether smaller river systems/estuaries have discrete stocks or if there is coastal mixing. In the absence of biological reference points for smelt, exploited stock status can only be assessed as trends in catch per unit of effort for the commercial fishery, trends in abundance in estuarine netting surveys (e.g. Coates *et al.*, 2007), or surveys of fish impingement on power station intake screens.

Belgium (Flanders)

In the early 1990s, smelt occurred in large numbers in the Zeeschelde Estuary (Vrielynck *et al.*, (2003). In 1991, a decline in numbers was observed (Rutgeers, 1912). Historically, spawning places were in the Nete River (Grote and Kleine; de Selys-Longchamps, 1842). In the beginning of the 20th century, the spawning grounds in the Nete were no longer used by smelt, which instead spawned in the Zeeschelde. From 1923 onward, industrial pollution resulted in the disappearance of smelt in the Zeeschelde. Other causes for the decline of smelt are the disappearance of low dynamic habitats, such as marshes and mudflats, essential to the reproduction of smelt. In 1990, smelt was captured again and is now the most abundant fish in the Zeeschelde Estuary. Adults are captured in spring, whereas in summer and autumn, large numbers of juveniles are caught. The spawning grounds are unknown.

Estonia

Annual monitoring data from River Pärnu and Pärnu Bay (adult spawner and larvae surveys). An annual gillnet survey is also conducted in Narva Bay in spring (Gulf of Finland). Fragmented data exist from River Pirita where a smolt trap is operating in spring.

Germany

Populations of smelt have been reported in German estuaries including the River Elbe (Scholle and Schuchardt, 2012). A Red Data Book for fish from German marine waters was published in 2013 (Thiel *et al.*, 2013).

Ireland

No national population estimate is available, but repeat surveys point to substantial spawning populations and recruitment in the Nore, Suir, and Shannon. Surveys have shown the presence of ripe male and female fish. Very small numbers of fish have also been taken in the Slaney Estuary since 2003. Sampling indicates established spawning populations and possible expansion of range into new waters (Slaney).

Netherlands

In Dutch waters, smelt migrate from the freshwater lakes IJsselmeer and Markermeer to the Wadden Sea, but there is no migration recorded from marine to freshwater (Phung *et al.*, 2015). In Lake IJsselmeer, different types of smelt have been found, based on otolith microchemistry, originating in local populations and from individuals that have spent time in Lake Markermeer and in tributaries such as the River IJssel.

Poland

There is a mass spawning migration from the Baltic, Vistula and Szczecin Lagoon to Vistula, Oder, and other rivers flowing into the sea in February–April.

Sweden

There is annual monitoring data and assessments from acoustic surveys in the larger lakes since 2008 (Axenrot and Beijer, 2015). In addition, smelt are regularly caught in monitoring coastal fishing in the Baltic Sea (HELCOM, 2012a).

UK (England and Wales)

Smelt occur in at least 36 watercourses in England and Wales, with large populations in the rivers Thames, Humber, and Dee, the Wash and Great Ouse, and in watercourses of the Norfolk Broads (e.g. Rivers Waveney, Yare, Bure, and Wensum). Smaller populations exist in the rivers Alde/Ore, Ribble, and Conwy, and recovery of supposedly extinct populations seems to be underway in the rivers Tyne and Mersey (Colclough and Coates, 2013). The status of smelt populations is uncertain, but can be considered vulnerable. Reviews by Maitland (2003) and Colclough and Coates (2013) indicate large populations in a number of river/estuaries, and reports of small numbers of smelt from other estuaries. Abundance is high enough to support commercial fishery in rivers and estuaries.

4.7.4 Genetics

In contrast to Holarctic salmoniform fish (e.g. coregonids, thymallids, and salmonids), the genetics of the Osmeridae have not been well studied. Most studies have examined the population genetics of the rainbow smelt (*Osmerus mordax*) and *O. m. dentex* over a range of spatial scales (e.g. Taylor and Bentzen, 1993; Bernatchez, 1997; Kovach *et al.*, 2013; Skurikhina *et al.*, 2015), or have examined the phylogenetic relationships of the genus *Osmerus*.

The genetic divergence between osmerid species within this genus has been examined using protein allozyme variation (Luey *et al.*, 1982), RFLP analyses of mitochondrial (mt) DNA segments and cytochrome b sequences (Taylor and Dodson, 1994), and, most recently, using RFLP analysis of mtDNA segments and sequencing of the cytb and COI genes (mtDNA) and intron 1 of the rpS7 gene (nDNA) (Kovpak *et al.*, 2011). Ocalcwicz *et al.* (2007) summarized that the genus *Osmerus* contains at least three species: European smelt (*O. eperlanus*) in northwest Europe, *O. mordax mordax* in the Northwest Atlantic, *O. mordax dentex* in the Arctic and North Asia, and *O. spectrum* occurring in lakes in eastern Canada and New England, USA. However, Bradford *et al.* (2013) grouped *O. m. mordax* and *O. spectrum* in Lake Utopia as two coexisting (sympatric) morphologically, ecologically, and genetically differentiated populations of the same species.

Recent molecular-based genetic phylogeographic and/or phylogenetic studies (Taylor and Dodson, 1994; Bernatchez, 1997) demonstrate that *O. eperlanus* and *O. mordax* split some 2.5–2.0 million years ago, and that the separation of *O. m. mordax* and *O. m. dentex*, and their spread along the coasts of both continents, occurred after the opening of the Bering Strait (Curry *et al.*, 2004). Available genetic data show a separation of *Osmerus* into clusters in accordance with their species affiliation, albeit with weaker subdivision into local populations depending on their geographic locality (Kovach *et al.*, 2013; Skurikhina *et al.*, 2015).

The number of genetic studies focusing on *Osmerus eperlanus* is limited. A search of Web of Science (WS) and Google Scholar (GS) (conducted September 17, 2016) using

the search term “*Osmerus eperlanus*” and scrolling through all WoS references and the first 20 pages on GS revealed two references on cytogenetics (Nygren *et al.*, 1971; Ocalwicz *et al.*, 2007) and two references on population genetics (Taylor *et al.*, 2008; Hagenlund *et al.*, 2015) for the European smelt. Nygren *et al.* (1971) and Ocalwicz *et al.* (2007) have investigated the cytogenetics of the European smelt, with Nygren *et al.* (1971) reporting that *O. eperlanus* has the lowest chromosome number reported among salmoniform fish. Ocalwicz *et al.* (2007) investigated the karyotype and cytogenetic characteristics of European smelt (*Osmerus eperlanus*) using different staining techniques to detect 5S rDNA and telomeric sites. Cytotaxonomic comparisons of their genome size data with that available for salmoniform and esociformes fish, as well as karyotype differentiation patterns, support the current view that salmoniform and osmeriform fish are not as closely related as previously assumed.

Taylor *et al.* (2008) isolated 12 dinucleotide microsatellite loci for *O. eprlanus* using smelt from the Forth (Scotland) and the Wadden Sea (The Netherlands), which provides a good basis for investigating *O. eperlanus* population structure. These microsatellite markers (plus the *O. mordax* locus OSMO-Lav 12; Taylor *et al.*, 2008) have been used by Tysklind *et al.* (in preparation) to study the population genetics of the European smelt using samples obtained from 11 anadromous populations (Cree, Forth, the Wash, Thames, Tamar in the UK; Scheldt, Belgium; Waddensee, The Netherlands; Weser and Eider, Germany; Curonian Lagoon, Lithuania; Gulf of Riga, Estonia) and one landlocked lacustrine population (Lake Peipsi, Estonia). The 13 loci showed between 6 and 27 alleles, with an average of 13.77 alleles per locus, and observed heterozygosity ranging from 0 (i.e. fixed for one allele) to 0.900, depending on the population. The programme STRUCTURE (Pritchard *et al.*, 2000) was used to estimate the number of separate populations sampled, using an admixture model, 10 000 iterations burn-in and 10 000 iterations sampling. This analysis indicated that the 12 populations clustered with the most likelihood into 9 separate groups. G tests for population differentiation on all population pairs indicated significant *p*-values for all pairwise population comparisons, except for Weser–Eider, which were not significantly different from each other. *F_{ST}* values (a measure of population differentiation) between population pairs for the 12 populations ranged between 0.007 (Weser–Eider) and 0.309 (the Wash–Lake Peipsi). These results largely agreed with those indicated by the STRUCTURE programme with the Cree, Forth, Wash, Curonian Lagoon, and Estonian (Gulf of Riga and Lake Peipsi) populations showing relatively high *F_{ST}* with most other populations, and lower *F_{ST}* values with closer populations. Both the Wadden Sea and Scheldt populations (the most central populations) were the most diverse and showed the least genetic differentiation from the other populations. This may be because they have the most potential for gene flow/migration to and from the other populations, being the most centrally located. The populations within the Baltic Sea were quite similar to each other and differentiated from those outside the Baltic

A recent wildlife forensics study used the molecular genetics of *O. eperlanus* to identify the source of a translocated population (Hagenlund *et al.*, 2015). European smelt were discovered to be present in Lake Storsjøen, south-central Norway, in 2008, despite no previous record in this lake. It was thought that this was the result of a translocation event from an unknown source, most likely through an intentional or accidental action when using smelt as fishing bait (Hagenlund *et al.*, 2015). The genetic variation in smelt from Lake Storsjøen was compared with several potential source populations using 13 microsatellite loci (9 *O. eperlanus* loci and 4 *O. mordax* loci). The results indicated that the most likely source was nearby Lake Mjøsa, supporting the initial hypothesis that

the translocation of smelt occurred from a single source in geographic proximity to Lake Storsjön (Hagenlund *et al.*, 2015).

4.7.5 International status with country highlights

International

OSPAR Convention:	Not listed
EC Habitats Directive:	Not listed
CITES:	Not listed
Bern Convention:	Appendix III (2002)
Bonn Convention:	Not listed
IUCN Criteria:	–
Global IUCN Red List Category:	LC
European IUCN Red List Category:	LC
HELCOM Red List:	European smelt (<i>Osmerus eperlanus</i>) - LC (HELCOM (2007) and HELCOM (2013a)) The subspecies <i>Osmerus eparalanomarinus</i> was considered VU in HELCOM (2007). However, the subspecies was not recognized as a separate species in the HELCOM checklist of macroscopic species (HELCOM, 2012a), and hence it was assessed as LC in 2013.

National IUCN Red List classifications (HELCOM Red List)

Estonia:	Minimum landing size (12 cm TL) / NT
Finland:	LC
Ireland:	LC (King <i>et al.</i> , 2011)
Poland:	No minimum landing size, fishing of smelt is allowed the entire year / not listed
Sweden:	LC
UK:	No formal assessment is made of smelt populations, although Maitland (1979) and Maitland and Lyle (1991) described the species as threatened because it has disappeared from many rivers over the last two centuries. However, Maitland (2003) reviewed information on smelt in England and concluded that thriving populations exist in several areas. Smelt are a Biodiversity Action Plan (BAP) priority species since 2007. UK BAP priority species are those that are identified as being the most threatened and requiring conservation action under the UK Biodiversity Action Plan. Maitland (2003) also covers Wales (includes Dee and Conwy). Maitland and Lyle (1996) reviewed smelt (sparling) in Scotland

The conservation status of smelt in different countries is shown in Table 4.7.

Table 4.7. Conservation by country of anadromous smelt.

Country	Criteria			
	National classification		International classification	
	2005 (ICES, 2005)	Last assessment	Habitats Directive	IUCN V 2015-4
Global			Annexes II, V	LC Population trend: unknown
Estonia	No category assigned	NT (Saat, 2008) - local red list		
Finland	No category assigned	LC (Urho et al., 2010)		LC
Ireland	VU (Whilde, 1993)	LC (King et al., 2011)	Not listed	
UK	No category assigned	VU (Maitland, 2000)		
Spain	EX	RE (Doadrio, 2001)		
Portugal	No category assigned	CR (Cabral et al., 2005)		

4.7.6 Distribution summary

In the North Atlantic the smelt distribution range extends from the White Sea in the north to the west coast of France in the south, including the Baltic Sea, southern North Sea, and British Isles. The Gironde Estuary is the southern limit of the historical distribution. Landlocked populations exist in coastal area lakes of the North, Baltic, White and Barents seas up to about latitude 68°N in the Scandinavia. The former nominal subspecies *Osmerus eperlanus eperlanus* is primarily lacustrine, but is recorded from the coasts and drainage of the White and Barents seas, and westward through the Baltic Sea to Denmark. The former nominal subspecies *Osmerus eperlanus schonfoldi* (Rutty, 1772) is sympatric, primarily anadromous and is found in parts of Denmark, Poland and the Baltic Sea.

In the Baltic Sea, smelt regularly reproduces in rivers and estuaries running to Kiel Bay, Bay of Mecklenburg, Arkona Basin, Bornholm Basin, Western Gotland Basin, Gulf of Riga, Northern Baltic Proper, Gulf of Finland, and Gulf of Bothnia (HELCOM, 2012a).

4.7.7 Country-by-country changes in distribution

A summary of changes in smelt distribution by country can be seen in Table 4.8.

Belgium

A decline in numbers of smelt was observed at the start of the 20th century (Rutgeers, 1912). Historically, spawning occurred in both branches of the Nete River (Grote Nete and Kleine Nete; de Selys-Longchamps, 1842). In the beginning of the 20th century, the spawning grounds in the Nete were no longer used by smelt, and spawning has not been recorded in the Nete in recent years. From 1923 onward, industrial pollution resulted in the disappearance of smelt in the Zeeschelde. Other causes included the disappearance of low dynamic habitats, such as marshes and mudflats, essential to the

reproduction of smelt. In the early 1990s, smelt occurred again in large numbers in the Zeeschelde Estuary (Vrielynck *et al.*, 2003), and has been captured in recent annual surveys. At present, smelt is the most abundant fish in the Zeeschelde Estuary. Adults are taken in spring. The location of the spawning areas has not been found, but large numbers of juveniles are caught in summer and autumn, confirming successful spawning in the Zeeschelde (Breine *et al.*, 2011a, 2011b).

Estonia

Smelt is most abundant in Pärnu Bay. Spawning runs occur into River Pärnu, but a small part of the population arguably also spawns in the bay. Small, brackish-water spawning populations also occur in Matsalu Bay and, probably, in Eru Bay. In the Gulf of Finland, anadromous runs occur in the rivers Pirita and Narva, and possibly also in some others. Mostly monitored in the Pärnu Bay and River Pärnu, but also in Narva Bay.

Finland

Smelt distribution in Finland covers many inland waters and nearly the entire coastal area, where they migrate to rivers or river mouths to spawn. The reproduction areas of smelt are mapped for the Finnish coastal area, and they cover nearly all estuaries or river mouths (Kallasvuo *et al.*, 2016).

Ireland

ICES 2005 report included coverage of the species for Ireland, summarized in Quigley *et al.* (2004), and flagged that populations of smelt had been encountered in several waters, but confirmed spawning had not been reported for all of these waters. Investigations on smelt status have been undertaken by Inland Fisheries Ireland (IFI) and its predecessor since the early 2000s. The presence of ripe adult fish at spawning time in the upper reaches of estuaries points to reproductive populations. This is supported by the occurrence of spent adult fish and of post-larval fish in bongo netting surveys, and of a series of size classes in autumn sampling¹⁶ (O' Gorman *et al.*, 2014).

The distribution of smelt in Ireland is particularly disparate, with original records from the Shannon Estuary, and subsequent ones from the Foyle and the Suir. Records of the type described above have been collected by IFI from the following waters:

- Foyle Estuary and tributary channels (working with Loughs Agency) – Foyle, Finn, and Dee (Niven, 2011)
- Shannon Estuary and tributary channels – Shannon, Fergus, Maigue.
- Munster Blackwater Estuary.
- Waterford Harbour – estuaries of the Suir, Nore, and Barrow.
- Slaney Estuary.

Sampling in spring (ca. mid-March) in the waters above, at the upper tidal limit using fykenets, has been successful in collecting samples of adult smelt. The only exception was on the Slaney where in two years spring sampling did not yield adult fish. As outlined, the present status confirms the distribution reported in 2005 regarding the Shannon, Foyle, and Suir, and expands the range in the large estuaries of the southeast. The surveys also indicate the presence of spawning populations. The occasional occurrence of individual fish of differing size classes in the Slaney Estuary suggests either

¹⁶ <https://www.fisheryireland.ie/Projects/habitats-directive-and-red-data-book-fish-species.html>

a very small spawning population or the opportunistic presence of fish. A smelt was identified from Bannow Bay (Karin Dubsky, pers. comm.). This water lies midway along the southeast coast between established smelt populations in Waterford Harbour and the isolated fish reported from the Slaney. The majority of Irish estuaries have now been sampled in autumn using a range of sampling gears, and smelt have only been recorded from the waters listed above. This supports the idea of a very disparate species distribution.

Latvia

Most abundant in the Gulf of Riga. The main spawning river in the Gulf of Riga is Lielupe, whereas in the main Baltic Sea it is river Venta.

Netherlands

Smelt is caught along the Dutch coast in estuaries such as Westernscheld and Ems-Dollard, and in the freshwater lakes IJsselmeer and Markermeer. It can also be found in Haringvliet and Nieuwe Waterweg and tributaries of these lakes, but not far inland. After closing the former Zuiderzee with dikes (Afsluitdijk in 1932 and Houtribdijk in 1976) landlocked populations of smelt were recorded in Lake IJsselmeer and in Lake Markermeer. In the northern lake, Lake IJsselmeer, there is a limited exchange of smelt with the Wadden Sea through the sluices in the Afsluitdijk. In the southern lake, Lake Markermeer, there is no exchange with marine water. The smelt population in Dutch waters shows a declining trend since monitoring started in the 1980s.

Poland

The distribution of smelt in Poland covers inland waters (landlocked populations) and almost the whole coastal area, from where they migrate to lagoons and rivers for spawning. Mass reproduction migration takes place from the Baltic Sea to the Vistula and Szczecin Lagoon, Vistula, Oder and smaller rivers flowing into the sea. The main migration period of migration is February to March (NMFRI, unpub. data; Heese, 1999).

Sweden

Smelt regularly occurs along the Swedish coast south to southern Östergötland. It is also widely distributed in larger lakes in central Sweden. Large populations are found in lakes Vänern, Vättern, Mälaren, and Hjälmaren. Some populations migrate into running water to spawn.

UK

As of 2015, smelt occur in at least 36 watercourses in England and Wales, with large populations in the rivers Thames, Humber, and Dee, the Wash and Great Ouse, and in watercourses of the Norfolk Broads (e.g. Rivers Waveney, Yare, Bure, and Wensum). Smaller populations exist in the rivers Alde/Ore, Ribble, and Conwy, and recovery of supposed extinct populations seems to be underway in the rivers Tyne and Mersey (Colclough and Coates, 2013). In Scotland, it is found in three rivers: Forth, Tay, and Cree. The Cree is the only river of 12 in southern Scotland (9 in Dumfries and Gallo-way) that previously had smelt populations (Maitland and Lyle, 1996).

Table 4.8. Distribution of anadromous smelt by country.

Country	River	Historic distribution	Reference	Present distribution	Reference	Trend direction
Portugal	Minho	X	Baldaque da Silva (1891)			
	Mondego	X	Baldaque da Silva (1891)			
	Tagus	X	Baldaque da Silva (1891)	X	Mateus <i>et al.</i> (2012, 2016)	Unknown
Belgium	Zeeschelde	Zeeschelede and Rupel	Poll (1947)	Zeeschelde, Rupel	Breine <i>et al.</i> (2011a, 2011b)	Improving since 2004
UK				36 waters in England and Wales; 3 in Scotland	Colclough and Coates (2013)	Improving
Ireland	Shannon and tributaries	X	Kennedy (1948)	X	Quigley <i>et al.</i> (2004)	
	Foyle and tributaries		Vickers (1974)	X	Niven (2010)	
	Barrow			X	O’Gorman <i>et al.</i> (2014)	
	Nore			X	O’Gorman <i>et al.</i> (2014)	
	Suir		Quigley (1996)	X	O’Gorman <i>et al.</i> (2014)	
	Munster Blackwater			X	O’Gorman <i>et al.</i> (2014)	
	Slaney			?	King and Linnane (2004)	
Finland	In several estuary areas	Not available	Not available	In several estuary areas	https://laji.fi/taxon/MX.53142	Unknown
Estonia	Pärnu	X	Špilev and Turovski (2003)		Špilev and Turovski (2003)	Increasing
Sweden	Narva	X			H. Špilev (unpub. data)	Unknown
Poland	Vistula and Vistula Lagoon			X	NMFRI (unpub. data)	Unknown
	Oder and Szczecin Lagoon			X	NMFRI (unpub. data)	Unknown
	Polish lagoon lakes			X	Heese (1999); NMFRI (unpub. data)	Unknown

4.7.8 Threats

Smelt is threatened by a variety of factors in different countries throughout its range.

Water quality

- Pollution in estuaries was implicated in the collapse of UK stocks. Where water quality has improved, populations have shown a recovery (Colclough and Coates, 2013).
- In Estonia, decreases in smelt abundance in the 1980s-1990s were associated with unfavourable natural spawning conditions and continuous pollution of River Pärnu. A subsequent marked reduction in agricultural and industrial pollution has resulted in increased smelt numbers.
- Some Swedish populations are threatened by increasing water temperature.
- The smelt stocks in the two Dutch inland seas have decreased dramatically over the years owing to less favourable water quality conditions, particularly in Lake Markermeer where the water is very turbid.

Barriers to migration/anadromy

The construction of large dams has adversely impacted smelt migration and spawning in some major catchments.

- In Estonia, the migration route on the River Pärnu (where the largest runs of anadromous smelt occur) is blocked by the Sindi Dam at river km 15 (the dam is currently (2019) being demolished).
- In Latvia, smelt abundance decreased after a hydropower plant construction in the lower part of the River Daugava (Aleksejevs and Birzaks, 2011).
- In Sweden, obstruction to migration has probably affected populations negatively, with reports of failed reproduction because of less well adapted water regulation schemes.
- Major fluctuations in water levels downstream of a major hydrodam in Ireland can lead to egg deposition on surfaces that subsequently become dried out, with a consequent loss of production (Quigley *et al.*, 2004).

Abstractions

All classes of smelt can be impacted by impingement onto intake screens at water abstraction facilities for industry supply and thermal power stations. Many such abstraction points occur in estuaries. Such impacts are reported from England (UK; Colclough and Coates, 2013) and from Northern Ireland (UK; Moorhead and Service, 1992; Niven, 2011; T. Harrison, unpub. data).

4.7.9 Exploitation

Smelt is continuously monitored only in few countries (Table 4.9).

Commercial fishing

Smelt is commercially exploited in a number of states bordering the Baltic Sea (see Heessen *et al.*, 2015). Some exploitation also occurs in UK. In Estonia, smelt is fished commercially, relatively heavily, in spring during spawning (especially in Pärnu Bay, but also in Narva Bay). Maximum landings were recorded in the 1960s-1970s when up to 2500 t of smelt were caught in the Gulf of Riga region alone. This was followed by a steep decline in smelt abundance to < 50 t per year. On from the turn of the present

century, smelt abundance has started to steadily increase, and up to 745 t of smelt have been caught yearly in the Pärnu Bay area in recent years. The minimum landing size is 12 cm (TL). Commercial exploitation also occurs in Latvia, where landings have decreased, and in Sweden, where heavy fishing pressure occurs in some areas, such as Lake Malaren, whereas other areas are not exploited.

Because of the negative impact on piscivorous birds in the lakes, the commercial fishery for smelt in both inland seas in the Netherlands has been closed in recent years (Heessen *et al.*, 2015).

An intense fishery existed in Belgium until 1910 (Vrielynck *et al.*, 2003), but there is no commercial fishing now.

In UK, smelt abundance is considered high enough to support commercial fishery in rivers and estuaries. These typically supply the catch for use as dead bait by anglers, although there is also a limited trade to restaurants. Since 2011, it has been a requirement for commercial fishery in England and Wales to be formally authorized by the Environment Agency/Natural Resources Wales. These authorizations also require the submission of annual catch returns. Reported catches were 3.2 t in 2011, but have been higher in more recent years (11.2 t in 2012, 14.2 t in 2013, and 11.0 t in 2014) (Environment Agency, 2012, 2013, 2014; Environment Agency and Natural Resources Wales, 2015). Continued analysis of catch reports is required, as are discussions with fishers about the spatial distribution of gears, and, ideally, the introduction of fishery-independent surveys of smelt abundance alongside the commercial fishery (higher priority) and in other waters (lower priority). The fishery operate in spring when smelt congregate for spawning, so there is a risk of overexploiting the spawning stock, as occurred in Estonia (details above).

There is no commercial fishing in Ireland. Small amounts are reportedly taken for private use as pike bait in Limerick during the spawning run on the River Shannon.

Recreational fishing

Smelt is a common target for anglers during the ice cover period in late winter and/or early spring in Estonia. It is used for human consumption (dried, fried). It is also popular for ice fishing in Latvia where annual landings from this activity could be 100–200 t. In Ireland, frozen smelt can be purchased in fishing tackle shops, and it is used as pike angling bait in some areas.

Table 4.9. Monitoring programmes and data for anadromous smelt.

Country	Monitoring programmes	Available data
Estonia	Annual monitoring data from River Pärnu and Pärnu Bay (adult spawner and larvae surveys). Annual gillnet survey is also conducted in Narva Bay in spring (Gulf of Finland). Fragmented data from River Pirita where a smolt trap is operating in spring.	Data collected as a part of the national fish monitoring programme. For Pärnu Bay, monitoring data available from late 1950s to the present day. Commercial landings available from late 1920s.
Ireland	Annual post-larval recruitment programme in estuaries of the southeast	https://www.fisheryireland.ie/Projects/habitats-directive-and-red-data-book-fish-species.html

4.7.10 Main conservation concerns

Smelt are highly susceptible to a range of environmental and anthropogenic impacts and are subject to large fluctuations in abundance (Hutchinson and Mills, 1987; Quigley *et al.*, 2004). Obstruction to migration has probably affected the populations negatively, and there are reports of failed reproduction as a result of less well-adapted regulation schemes. Climate change and sea temperature rise are considered to be the main threats to smelt. Spawning populations are vulnerable to pollution events. Sediment deposition on spawning grounds reduces spawning habitat.

In Estonia, decreases in smelt abundance in the 1980s–1990s were associated with unfavourable natural spawning conditions and continuous pollution of River Pärnu. At the turn of the century, agricultural and industrial pollution decreased markedly, and this has resulted in an increase in smelt numbers, although not yet to historical levels. In England and Wales (UK), pollution in estuaries was implicated in the collapse of stocks, but populations have shown a recovery where water quality has improved (Colclough and Coates, 2013).

In some areas (e.g. Estonia, River Pärnu), the migration routes are blocked with dams, which can notably reduce the suitable reproduction area. Smelt are relatively weak swimmers and, therefore, access from estuaries into freshwater for spawning can be disrupted by weirs and other structures. Even major riffles can be barriers to migration. As a consequence, bypass devices designed to assist other diadromous fish in overcoming physical barriers are unlikely to be effective for smelt. Smelt are also susceptible to entrainment or impingement on power plant intake screens.

Smelt are considered to be vulnerable to exploitation, although regional differences are large. In some areas, smelt is heavily fished. No controls on exploitation exist except in Estonia, where the minimum landing size is 12 cm (TL). The fishery operate in spring when smelt congregate for spawning, so there is risk of overexploiting the spawning stock.

4.7.11 Conclusion – future

It is clear that smelt face different pressures throughout their range, with heavy commercial fishery in Baltic states and reduced, or zero, commercial fishing in other areas. There are pressures on populations if pollution and fish kill events occur, and physical barriers to migration remain an issue.

Smelt has a demonstrated plasticity, with both freshwater and anadromous forms. It has also shown the capacity to colonize or recolonize areas when water quality conditions have improved.

Some national monitoring programmes are in place for smelt. In other cases, surveillance monitoring programmes of estuarine and lake waters under the EU Water Framework Directive will provide a database for assessing smelt populations (as well as other fish species) in the future. Smelt are a sensitive indicator of good ecological status for transitional waters under the Water Framework Directive.

In Ireland, sampling indicates established spawning populations and a possible range expansion into new water (Slaney). Repeat surveys point to substantial spawning populations in the Nore, Suir, and Shannon.

In Estonia, annual monitoring data exist from River Pärnu and Pärnu Bay (adult spawner and larvae surveys). There is an annual gillnet survey in Narva Bay in spring (Gulf of Finland) and fragmented data from River Pirita, where a smolt trap is operating in spring.

Annual monitoring data and assessment from acoustic surveys in the larger Swedish lakes is available since 2008 (Axenrot and Beijer, 2015). Monitoring of coastal fishing in the Baltic Sea also produces data on smelt status (HELCOM, 2012b).

Recent data from Belgium show that the number of individuals has increased in recent years.

Smelt culture for restocking purposes

Removing the threats causing population decline, in order to promote habitat improvement and allow the species to recover naturally, is the primary means by which declining populations of threatened fish can recover. However, it is sometimes necessary to use stock enhancement to help promote recovery. Conservation aquaculture involves the development of aquaculture rearing techniques in order to conserve or aid the recovery of threatened fish populations. Techniques for captive rearing and culture of rainbow smelt (*Osmerus mordax*) for restocking and baitfish culture in New England, USA, were developed 10 years ago (e.g. Akielaszek *et al.*, 1985; Ayer *et al.*, 2005; Fuda *et al.*, 2007; Colborn *et al.*, 2012). In contrast, the captive rearing of European smelt through its full life cycle from egg to mature adult was only developed recently by Jones and McCarthy (2013), showing that it is possible to rear European smelt in captivity for stock enhancement programmes. Prior to that, only the small-scale culture of eggs and larvae of *Osmerus eperlanus* in petri dishes to study embryogenesis and larval development had been conducted (Gorodilov and Melnikova, 2006a, 2006b).

In their study, Jones and McCarthy (2013) obtained fertilized eggs from adults returning to spawn in the River Cree at Newton Stewart, southwest Scotland (Hutchinson and Mills, 1987). Fish were reared at Bangor University for two years until mature adult fish with ripe, running gonads were obtained. Experimental work indicated that the optimum rearing conditions to maximize hatching success (90–100%) and larval survival to first-feeding (92–100%) were to rear eggs at 5–10 °C and at salinities of 0–10 psu (Jones and McCarthy, 2013). Juvenile European smelt were successfully weaned from live feed (rotifers and *Artemia*) to commercial pelleted feed, but this life stage was susceptible to stress and suffered mortalities if handled (Jones and McCarthy, 2013). This work has shown that European smelt can be cultured for restocking purposes. However, it is recommended that fish are restocked into estuaries at the larval phase rather than growing to juveniles. The aim of captive culture should be to maximize survival through to the hatched yolk sac larval stage, and to restock at an early life stage, before culture can negatively impact anti-predator and feeding behaviours.

4.8 Three-spined and nine-spined stickleback

4.8.1 Taxonomy

Class:	Actinopterygii
Order:	Gasterosteiformes
Family:	Gasterosteidae
Scientific name:	Three-spined stickleback - <i>Gasterosteus aculeatus</i> Nine-spined stickleback - <i>Pungitius pungitius</i>

Subspecies, variations, synonyms, and taxonomic notes

Gasterosteus gymnurus (Cuvier, 1829).

There is a hybrid zone with *G. gymnurus* in the English Channel, southern North Sea, Baltic Sea, and their basins (Hammerson *et al.*, 2010). “*Gasterosteus aculeatus*” In IUCN (2014). IUCN Red List of Threatened Species. Version 2014.1.

Gasterosteus leiurus (Cuvier, 1829).

Preliminary observations suggest that several of the isolated populations in the Mediterranean basin should possibly be recognized as distinct species (Freyhof and Kottelat, 2008). *Gasterosteus gymnurus*. The IUCN Red List of Threatened Species, 2008).

Gasterosteus semiarmatus (Cuvier, 1829).

Gasterosteus semiloricatus (Cuvier, 1829).

Gasterosteus tetracanthus (Cuvier, 1829).

Gasterosteus brachycentrus (Cuvier, 1829).

Gasterosteus nemausensis (Crespon, 1844).

Gasterosteus quadrispinosa (Crespon, 1844).

Gasterosteus argentatissimus (Blanchard, 1866).

Gasterosteus bailloni (Blanchard, 1866).

Gasterosteus elegans (Blanchard, 1866).

Gasterosteus neustrianus (Blanchard, 1866).

Gasterosteus hologymnus (Regan, 1909).

Gasterosteus aculeatus messinicus (Stephanidis, 1971).

Common names

ET: ogalik/luukarits; ES: espinoso; GL (Galician): espiñento; CA (Catalan): Espinós; EU (Basque): Arrain hiruarantza; FI: Kolmipiikki, kymmenpiikki; PL: ciernik; SV: Storspigg, småspigg; PT: Esgana-gata.

General characteristics

Length (common):	Three-spined stickleback – 5.1 cm Nine-spined stickleback – 6.5 cm
Maximum recorded age:	Three-spined stickleback – 8 years Nine-spined stickleback – 5 years

The three-spined and nine spined sticklebacks are widely distributed across the Arctic and temperate coastal-zones, with populations also in inland waters. The species show variations in morphology, especially in the number of bony plates, which differs between populations (Froese and Pauly, 2015). In lakes, both limnetic and benthic forms occur (Schluter, 1993). These are usually reproductively isolated.

4.8.2 Life cycle and migrations

Anadromous populations co-occur with numerous non-anadromous populations in brackish or pure freshwater. Stickleback conduct offshore migrations in large schools to feed during the first two years of their life, but return to coastal areas or rivers for spawning (Bergström *et al.*, 2015). When males mature, at 1–2 years old, they become territorial, build a nest, and take care of the eggs (Froese and Pauly, 2015). The majority of the spawners die after reproduction. In many coastal areas of the Baltic Sea, sticklebacks have important roles in the foodweb. They are mesopredators that control the status and structure of zoobenthos, macro algal, and ephemeral algal communities (Eriksson *et al.*, 2011; Sieben *et al.*, 2011; Bergström *et al.*, 2015; Byström *et al.*, 2015; Östman *et al.*, 2016).

4.8.3 Stock structure and population dynamics

No data are available for the Baltic Sea area.

4.8.4 Genetics

There are no major genetic barriers, no isolation by distance pattern, and a low genetic differentiation within the Baltic Sea for both three-spined and nine-spined stickleback, although there is a higher differentiation among populations within the latter (Wennerström *et al.*, 2013). Recent molecular analysis of European samples of three-spined sticklebacks reported that populations from the Iberian Peninsula showed a higher level of divergence, extended periods of isolation, and ancient evolutionary histories (Araguas *et al.*, 2012; Sanz *et al.*, 2015). Evidence of isolation between Iberian subpopulations are exacerbated by small population sizes, which are considered to be below the minimum suggested to maintain adaptive genetic variation.

4.8.5 International status with country highlights

International

Both species are considered abundant, widely distributed, and not threatened in all countries in the previous report (ICES, 2005). They are not listed on any global conventions.

OSPAR Convention:	Not listed
EC Habitats Directive:	Not listed
CITES:	Not listed
Bern Convention:	Not listed
Bonn Convention:	Not listed
IUCN Criteria:	-
Global IUCN Red List Category:	LC (IUCN, 2008, 2013)
European IUCN Red List Category:	LC (IUCN, 2013, 2015)

HELCOM Red List: LC (HELCOM, 2013a)

National

Current country status are highlighted below and in tables 4.10 and 4.11:

- Estonia: Sticklebacks are currently not targeted by commercial fishers, but were historically exploited.
- Italy: Three-spined stickleback might be threatened (Elliot and Hemingway, 2002). However, the species is introduced in this area according to NatureServe (2015).
- Portugal: Three-spined stickleback is classified as EN by the Portuguese Red Book of Threatened Vertebrates (Cabral *et al.*, 2005), and its populations have shown accentuated declines in abundance over the last 10 years.
- Sweden: Commercial coastal and offshore fishery in 2015 landed 80 t of sticklebacks. No special legislation exists for this species.

Table 4.10. Conservation by country for the three-spined stickleback.

Country	Criteria		
	National classification		International classification
	2005 (ICES, 2005)	Last assessment	IUCN 2015
Global			LC
Baltic Sea			LC (HELCOM, 2013a)
Belgium	LC	LC (Verreycken <i>et al.</i> , 2013)	LC
Estonia	Na	LC	
Portugal		EN (Cabral <i>et al.</i> , 2005)	
Sweden	LC	LC (Artdatabanken, 2015)	

Table 4.11. Conservation by country for the nine-spined stickleback.

Country	Criteria		
	National classification		International classification
	2005 (ICES, 2005)	Last assessment	IUCN 2013
Global			LC
Baltic Sea			LC (HELCOM, 2013a)
Sweden	LC	LC (Artdatabanken, 2015)	
Estonia	na	LC	
Belgium (Flanders)	LC	LC (Verreycken <i>et al.</i> , 2013)	LC

Table 4.11 (continued)

Spain	<p>IUCN (2001): Endangered EN B1 + 2 abcde</p> <p>Regional legislation:</p> <p>EN in Annex 1 Regional Catalogue of Endangered Species of Extremadura, Law 8/1998 of 26 June.</p> <p>"Endangered Extinction" in the Basque catalogue of threatened species of fauna and wild and marine flora</p> <p>Decree 167/1996 of 9 July. s D in Annex II of protected wildlife species native, law animal protection 3/88 of Catalonia.</p> <p>VU in the Galician Catalogue of threatened species (regulated by Decree 88/2007).</p>
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4.8.6 Distribution summary

The three-spined stickleback has a wide, circum-Arctic and temperate distribution, and also occurs in the Black Sea (Figure 4.8; Froese and Pauly, 2016). The nine-spined stickleback has a similar wide, circum-Arctic distribution (Figure 4.9).

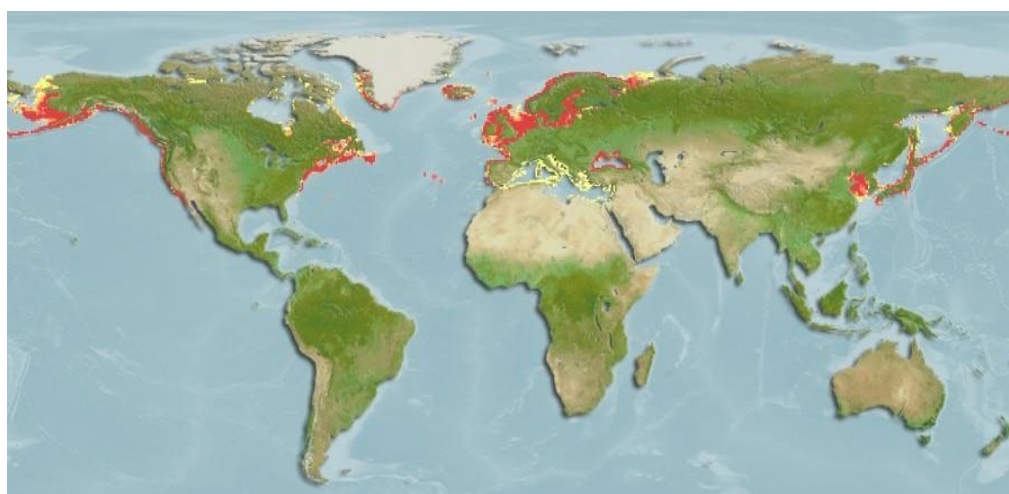


Figure 4.8. Distribution of three-spined stickleback. Reviewed distribution maps for *Gasterosteus aculeatus* (three-spined stickleback). Source: www.aquamaps.org, version of Aug. 2013. Web. Accessed 29 May. 2016).

4.8.7 Country-by-country changes in distribution

The abundance of three-spined stickleback has increased drastically in the western parts of the central Baltic Sea and Bothnian Sea during the last 20 years: fourfold in the Bothnian Sea, 45-fold in the central Baltic Sea, and sevenfold in the southern Baltic Sea (Bergström *et al.*, 2015). There are indications of a decline in sticklebacks in the Gulf of Riga (Olsson *et al.*, 2015). In Flanders (Belgium), the three-spined stickleback is a common fish in rivers and lakes. In estuaries, it is found up to the mesohaline zone. The nine-spined stickleback is not that abundant.

In contrast, in the Iberian Peninsula, three-spined stickleback populations have shown dramatic declines in recent years. In Spain, the species is currently experiencing a regressive tendency, and only occurs as some isolated populations located in small rivers in the Galicia, Catalonia, and Cantabria regions. In Portugal, the species' situation appears to be equally dramatic. The species can be detected in the majority of Portuguese river basins from Minho in the north to Mira in the south, but effective populations can only be found in some specific and isolated locations, and in reduced abundance.

No distribution change data exists on the nine-spined stickleback.

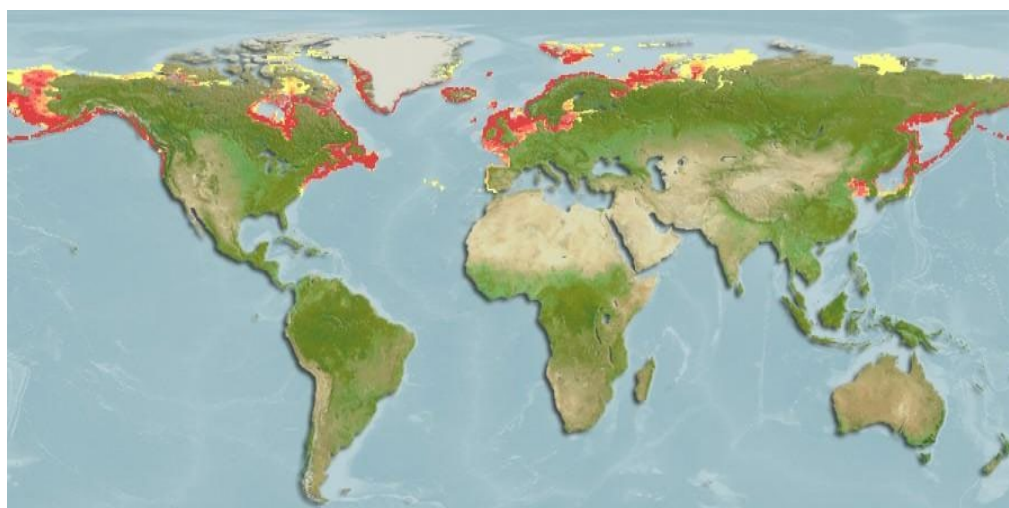


Figure 4.9. Distribution of nine-spined stickleback. Computer generated distribution maps for *Pungitius pungitius* (nine-spined stickleback), with modelled year 2100 native range map based on IPCC A2 emissions scenario. Source: www.aquamaps.org, version of Aug. 2013. Web. Accessed 29 May. 2016.

4.8.8 Threats

There are no known global threats for either species (NatureServe, 2015). However, for southern European populations, anthropogenic pressures related to water abstraction and habitat modification, which exacerbate the typical annual drying pattern of Mediterranean rivers, seem to be one of the main causes for the decline of three-spined sticklebacks (Alexandre and Almeida, 2009).

4.8.9 Conclusion – future

Management recommendations:

- The impact of the increase in the three-spined stickleback on other species should be investigated in the Baltic Sea and central Europe. There is evidence for a negative impact of sticklebacks on coastal predatory fish (Bergström *et al.*, 2015; Byström *et al.*, 2015) and on zoobenthos (Eriksson *et al.*, 2011; Sieben *et al.*, 2011), ultimately fostering blooms of ephemeral algae and decreased macroalgal densities (Eriksson *et al.*, 2011; Sieben *et al.*, 2011; Östman *et al.*, 2015).
- A comprehensive assessment of stickleback status and the definition of priority conservation should be carried out for southern European populations, owing to their specific taxonomic and conservation situation.
- In all areas, regular monitoring programmes should be initiated. Current monitoring of sticklebacks are shown in Table 4.12.

Table 4.12. Monitoring programmes and data in some countries of occurrence for sticklebacks.

Country	Monitoring programmes	Available data
Belgium (Flanders)	Schelde Estuary yearly monitoring with fykenets and anchornet (three seasons). Lakes, rivers, and canals: three yearly monitoring according to WFD with electric fishing and fykenets.	VIS databank (Brosens <i>et al.</i> , 2015)
Finland	1) Whitefish beach seining in the Quark area and in the northern Gulf of Bothnia 2) Echosounding survey of the Baltic herring stock in the Finnish sea areas as a part of EU's data collection programme	Sticklebacks are collected as bycatch from these two projects, from beach seining since the 1990s, from open sea surveys of herring during some recent years. Both studies give relative data on the number of sticklebacks and also length data.
Estonia	Sticklebacks are not monitored regularly. Only occasional monitoring during specific projects	No constant dataserries.
Portugal	Information on three-spined stickleback distribution, abundance, and population status is limited to occasional non-specific monitoring programmes and one scientific publication (Alexandre and Almeida, 2009).	No constant dataserries. Data on summer survival and habitat selection for a population inhabiting a temporary river in southern Portugal (Alexandre and Almeida, 2009). Increasing need for a comprehensive assessment of species distribution, abundance, and population status.
Spain	No monitoring programmes, but three spined stickleback known to have a wide distribution in Spain. Present in small enclaves in the basins of Galicia (where the species is more numerous and widely distributed), Cantabrian, Levante, Catalonia, and Mallorca. It seems extinct in the Guadiana	Data source - general: (Doadrio <i>et al.</i> , 2011), morphological variability (Lobón-Cerviá <i>et al.</i> , 1988), morphometric and meristic characteristics (Fernández <i>et al.</i> , 2000; Hermida <i>et al.</i> , 2005a), general ecology (Clavero <i>et al.</i> , 2009), biogeography (Hernando and Soriguer, 1992; Doadrio, 2001; Filipe <i>et al.</i> , 2009), fluctuating asymmetry (Hermida <i>et al.</i> , 2005b), and feeding habits and feeding behaviour (Sánchez-Hernández <i>et al.</i> , 2011; 2012)
Sweden	Coastal fish monitoring programmes (three-spined stickleback)	Several dataserries (>10) in Bothnian Sea and the Baltic Sea proper, oldest started in 1987, but most started in early 2000s. (HELCOM, 2012a; Bergström <i>et al.</i> , 2015).
	Surveillance in nuclear power plants (both species)	Data from Kattegat, Bothnian Sea, and central Baltic Sea cooling water intake since 1980s.
	Baltic International Acoustic Survey (three-spined stickleback)	Data from interational survey from 1970s to present (Bergström <i>et al.</i> , 2015)

4.9 Vendace

4.9.1 Taxonomy

Class: Actinopterygii
Order: Salmoniformes
Family: Salmonidae
Scientific name: *Coregonus albula*

Subspecies, variations, synonyms

None

Taxonomic notes

None

Common names

DE: Kleine Maräne; DA: Heltling ; ET: Rääbis; FI: Muikku; LT: Seliava; PL: Sielawa;
RU: Европейская ряпушка; SV: Siklöja

General characteristics

Average length: 15-20 cm
Average weight 0.020-0.030 kg
Average maximum age: 5-6 years

Vendace is a small, pelagic, schooling, freshwater fish, commonly occurring in lakes. An anadromous form is also found in the Gulf of Finland and in the brackish water of the Gulf of Bothnia. Vendace is subjected to a local commercial fishery in both Sweden and Finland. In Sweden, it is fished in autumn mainly for its roe, whereas in Finland it is especially caught for the consumption of its flesh.

4.9.2 Life cycle and migrations

Vendace reach sexually maturity-at-age 1–3 (Havs och vattenmyndigheten, 2015), but most individuals are mature at the age of one year. Spawning time is late autumn (October–December), when they migrate to spawn into the estuaries and lower reaches of rivers or into less saline coastal habitats, such as the Gulf of Bothnia (Veneranta *et al.*, 2013a, 2013b). Vendace spawn mainly at 3–10 m depth, or just below surface on shallow sand or gravel substrates. It feeds predominantly on zooplankton (Hamrin, 1983).

4.9.3 Stock structure and population dynamics

Swedish assessment data show that the spawning-stock biomass in the Swedish part of the Bothnian Bay has increased over the last 6 years (Havs och vattenmyndigheten, 2015). Landings have increased in line with the biomass, reaching > 1000 t annually since 2011. Finnish landings are substantially lower. In 2015, the total catch from commercial fishing was 132 t, and all vendace were caught in the Gulf of Bothnia. The vendace stock status in the Gulf of Finland is unknown, but stock size is assumed to be very low, based on the lack of vendace in the Finnish official catch statistics for the last 30 years.

Vendace show large annual fluctuations in recruitment, correlated mainly to variations in salinity and temperature, and also to fishing (Bergenius *et al.*, 2013). Year-class

strength is, therefore, highly variable from year-to-year. Recruitment is strongly connected to the fishable and mature part of the population in the coming years. Seal consumption of vendace has also increased as the ringed seal (*Pusa hispida botnica*) population in the Bothnian Bay has recovered from the intensive hunt pressure and reproduction problems in the 20th century. The yearly consumption of vendace by ringed seals is estimated to be at least the same size as the commercial landings (Lundström *et al.*, 2014), and is, therefore, likely to be affecting the size of the vendace population in the Bothnian Bay.

4.9.4 Genetics

Not much is known about the genetics of vendace. This topic will need more study effort in the future.

4.9.5 International status with country highlights

International

OSPAR Convention:	Not listed
EC Habitats Directive:	Not listed in appendix II or IV (2007) <i>Coregonus</i> spp. in Annex V
CITES:	Not listed
Bern Convention:	Appendix III
Bonn Convention:	Not listed
IUCN Criteria:	–
Global IUCN Red List Category:	LC
European IUCN Red List Category:	LC
HELCOM Red List:	LC

The size of the population and the extent of occurrence, as well as the area of occupancy, are well above the thresholds for being red listed according to the IUCN criteria. It is frequently stocked in lakes and reservoirs in northern and central Germany and Poland, as well as in Estonia. Baltic Sea and lacustrine populations are commercially fished in Sweden, Finland, and the Russian Federation, and its roe is very valuable.

Past and current threats: –

Future threats: –

National IUCN Red List classifications (HELCOM Red List)

Denmark	– / VU in freshwater
Estonia:	– / DD
Finland:	– / LC
Germany:	– / NT (freshwater)
Latvia:	– / RE (freshwater)
Lithuania	– / –
Poland:	– / DD

Russian Federation – / –

Sweden: Local, stakeholder managed fishery for a restricted number of license holders / LC

4.9.6 Distribution summary

Europe: Baltic basin, lakes of upper Volga drainage (Seliger, Vseluga, Perejaslavskoe), some lakes of White Sea basin, and North Sea basin east of Elbe drainage. Anadromous in Gulf of Finland, and marine in the northernmost brackish part of Gulf of Bothnia; north to about 69°N in Lake Inari (not native, stocked origin), northern Finland; lower Rhine (now extirpated). Frequently stocked in lakes and reservoirs in Germany and Poland (Figure 4.10).



Figure 4.10. Reviewed distribution maps for *Coregonus albula* (vendace) in coastal areas, www.aquamaps.org, version Aug. 2013. Web. Accessed 26 Apr. 2016.

4.9.7 Country-by-country changes in distribution

The big difference in catches between Finland and Sweden is mainly owing to the geographical and morphological differences in sea and coastal areas. In Sweden, the coast is quite deep compared to Finland. Vendace feed in the shallow Finnish coast during summer, when the sea is warmer than in Sweden. However, the main spawning areas are on the Swedish coast, where large vendace shoals gather to spawn in late autumn. Migrations between the Swedish spawning grounds and the Finnish feeding areas were shown by Lehtonen and Enderlein (1984). The main trawl fishing season in Sweden occurs during this spawning time when vendace are particularly vulnerable to capture. Presently, the stock in northern Gulf of Bothnia is harvested sustainably, and no excessive harvesting is recorded for other stocks in the main distribution area. This species is considered LC in the HELCOM area, and no protection actions are currently needed. Fishery statistics, however, should continue to be monitored to prevent over-fishing.

4.9.8 Threats

Vendace is a short-lived species with highly variable year classes. In the brackish Bothnian Bay, it lives at the margins of its distribution. Recruitment is highly influenced by small fluctuations in environmental factors such as salinity and temperature (Bergénius *et al.*, 2013). Climate change and its consequences is, therefore, the most likely threat to the marginal vendace population in the northern Gulf of Bothnia. Fishing, if not strictly managed, may pose a threat to vendace populations, because vendace is a highly sought-after fish in some areas on account of its exclusive roe. The recovered seal stocks and increasing cormorant populations are an additional risk factor. However, because of its short generation cycle, vendace usually recovers quickly from over-exploitation if the spawning grounds are in good condition and natural circumstances favourable.

4.9.9 Conclusion – future

To maintain sustainable fishery of anadromous and marine vendace populations throughout their distributions, fishery regulations, and fishery-dependent and -independent data should continue to be collected and used. In the Bothnian Bay, fishers play an active role in the management of vendace by collecting biological and fishery information, and through dialogue with researchers and managers. Such co-management contributed to the recovery of the population in the 1990s, and its continuation is highly recommended. The fishery on this marginal population is largely recruitment-driven, and recruitment, in turn, is sensitive to environmental variables and fishing. The successful management of this species consequently requires precautionary measures, taking into account both short- and longer-term changes in environmental variables.

4.10 Vimba bream

4.10.1 Taxonomy

Class: Actinopterygii
Order: Cypriniformes
Family: Cyprinidae
Scientific name: *Vimba vimba*

Subspecies, variations, synonyms

None

Taxonomic notes

None

Common names

DE: Zährte; DA: Vimme; ET: Vimb; FI: Vimpa; LV: Vimba; LT: Žiobris; PL: Certa;
RU: Сырть; SV: Vimma

General characteristics

Maximum total length: 50 cm
Maximum weight: 1.4 kg
Maximum recorded age: 16 years
Usual generation time: 6–10 years
IUCN Criteria: LC

4.10.2 General overview

Vimba bream are freshwater fish that also inhabit the brackish Baltic Sea (Figure 4.11, Table 4.13). Subspecies also occur in the Caspian and Black seas. All vimba bream populations in the Baltic Sea are obligatorily anadromous. This means that vimba bream is the most anadromous of the species that are regarded as freshwater species inhabiting the Baltic Sea.



Figure 4.11. Distribution of vimba bream (Freyhof and Kottelat, 2008e).

Table 4.13. Distribution of anadromous vimba bream in the Baltic Sea by country.

Country	Distribution
Denmark	Data not available.
Estonia	Very common in the Pärnu Bay area and in the Väinameri Sea region. Less common in the Gulf of Finland. In total, ca. 20 rivers are known as anadromous vimba bream spawning rivers, but the current situation is largely unknown, especially in the rivers that flow into the Gulf of Finland.
Finland	Anadromous vimba bream spawn in ca. 30 rivers (no freshwater resident populations). Current situation is largely unknown.
Germany	Anadromous vimba spawn in one or two rivers in the Oder Bay region. Vimba nearly disappeared in the 1970s, but have been recovering during the last 20 years. Vimba is endangered in the German Baltic area.
Latvia	Common in coastal waters; spawns in all the largest rivers. Distribution area has decreased since 1970 on account of damming in River Daugava. Historically found in ca. 63 rivers, but currently only in ca. 28 rivers (Birzaks <i>et al.</i> , 2011).
Lithuania	Common in Lithuanian coastal waters. Young vimba bream are often caught in the Curonian Lagoon during summer. Adults mainly start their spawning migration in autumn, and, in winter, in lower reaches of rivers. For spawning, vimba migrate up to at least 400 km upstream in the Nemunas River basin to reach Vilnius in Neris River or even the border with Belorussia. Vimba used to migrate upstream above Kaunas City until a hydropower dam was built in 1959. After the construction of the dam, a landlocked population of self-sustaining vimba bream exists above the dam in Nemunas River.
Poland	Present in Oder, Rega, and Vistula rivers. All those populations are supported by stocking. The building of a large dam in Wloclawek (Vistula River) in 1970 efficiently stopped the migrations of vimba bream. However, currently the population is recovering.
Russian Federation	Data not available.
Sweden	Likely found in many coastal areas, but limited information exists.

Maturity is generally reached at 6–8 years of age. All populations undertake spawning runs to larger rivers, and natal homing seems to be pronounced. Migrations in the sea and freshwater are extensive, and may reach up to 300–400 km in rivers (Erm *et al.*, 2003; L. Lozys, pers. comm.). Young-of-year probably descend to the sea in autumn. In the Baltic Sea, the most abundant vimba bream stocks probably occur in the Gulf of Riga, where they are heavily fished by Estonian and Latvian commercial and recreational fishers. Historically there was also an important fishery in Finland. Different spawning stocks mix quite extensively outside the spawning season (e.g. vimba bream from Latvian rivers feed and/or overwinter in Pärnu Bay and the Väinameri Sea). In Lithuania, commercial fishers have landed an average of ca. 90 t annually during the last decade. Vimba bream are also exploited by recreational fishers, but mostly in rivers. The stock in Lithuania was in decline during 1980–1999, but since 2000 it is regarded as stable after recovery (L. Lozys, pers. comm.). In general, some vimba bream spawning stocks in the Baltic Sea are currently viable, but they are low in number compared to historical levels; and other spawning stocks are declining or have been lost. Damming of rivers has heavily deteriorated the Gulf of Riga and Finnish stocks. Overfishing, dredging of rapids, and pollution of rivers have also negatively influenced vimba bream populations.

Existing monitoring programmes for vimba bream can be seen in Table 4.14.

Management recommendations:

- As a first step, dams should be removed from rivers where feasible.
- It is necessary to increase our knowledge of life history and ecology to suggest meaningful action plans.

Table 4.14. Monitoring programmes and data for anadromous vimba bream in the Baltic Sea countries.

Country	Monitoring and data
Denmark	Data not available.
Estonia	Monitored annually in several sea areas and in River Pärnu as a part of the national coastal sea fish monitoring programme; official landings from the sea.
Finland	Monitored as part of the national coastal sea fish monitoring programme; official landings from the sea.
Germany	No special monitoring for vimba; there are electrofishing surveys in the rivers (EU-water frame directive) and a trawl survey in the Pomeranian Bay of the Baltic Sea.
Latvia	Monitored in coastal waters and rivers; landings data from coastal waters and rivers; angling data from few rivers.
Lithuania	Monitored annually in several areas in the Curonian Lagoon and coastal waters of the Baltic Sea; official landings are available from the the Curonian Lagoon and coastal waters of the Baltic Sea.
Poland	No monitoring. Occasionally recorded during conducting of the monitoring within Multiannual Programme for Collection of Fisheries Data.
Russia Federation	Data not available.
Sweden	Occasionally caught in gillnet monitoring programmes.

4.11 Whitefish

4.11.1 Taxonomy

Class: Actinopterygii
Order: Salmoniformes
Family: Salmonidae
Scientific name: *Coregonus lavaretus*

Subspecies, variations, synonyms

Coregonus maraena
Coregonus balticus
Coregonus oxyrinchus
Coregonus pallasii

Taxonomic notes

In early 20th century, the name *C. lavaretus* was used for all Eurasian coregonids, and many North American ones. Its use is now restricted to the present species.

Common names

DE: Schnäpel; DA: Hetling; EE: Merisiig; FI: Siika; LT: Šiga; LI: Sykas; PL: Sieja; RU: Sig; SV: Sik

General characteristics

Length: 25–70 cm
Weight: 0.2–12 kg
Maximum age: 30 years
Generation length: 9 years

European whitefish are widely distributed across the Palearctic, both in inland waters and in coastal areas of the Baltic Sea (Froese and Pauly, 2015). The species shows great variation in morphology and ecological plasticity (Østbye *et al.*, 2005, 2006; Säisä *et al.*, 2008). In the Baltic Sea, two major ecotypes of whitefish are recognized in the coastal areas: a migratory form spawning in coastal rivers and creeks (hereafter referred to as river spawning ecotype/form), and a more resident form spawning in the coastal area of the Baltic Sea (hereafter denoted as sea spawning ecotype/form) (Säisä *et al.*, 2008; Olsson *et al.*, 2012). Both forms occur in coastal waters from the very north to more southern parts of the Baltic Sea. The coastal whitefish have nearly identical appearance (Himberg *et al.*, 2015). The taxonomy of whitefish is under debate and includes several species with different names (Kottelat and Freyhof, 2007).

Both whitefish ecotypes are important fish for coastal recreational and commercial fishery, especially in the northern Baltic Sea (Verliin *et al.*, 2013; OSF, 2015; Havs och Vattenmyndigheten, 2016).

4.11.2 Life cycle and migrations

Whitefish is a cold-water species that lives mostly in coastal waters. In the more saline southern Baltic Sea some populations occur close to estuaries. In the northern Baltic Sea, the river-spawning whitefish forages along coasts and carries out long migrations of 50–700 km (Lehtonen, 1981; Leskelä *et al.*, 2004). The northern-most stocks undertake

the longest migrations between spawning and foraging grounds. Whitefish in coastal areas feed mostly on benthic prey (crustaceans, molluscs, large insect larvae, and small fish; Verliin *et al.*, 2011), although no exact data are available for the entire distribution. River-spawning ecotypes migrate into freshwater for spawning in October–December (Lehtonen, 1981). Adults ascend rivers mainly during June–October (northern Baltic) or November (southern Baltic). Spawning occurs often in the lower reaches of rivers or rapids.

The sea-spawning ecotype is recognized to be more local, with shorter migrations (Olsson *et al.*, 2012). Sea spawners occur in subpopulations, and do not tend to migrate into rivers. Spawning takes place during October–December in various shallow coastal areas, such as estuaries or open shores (Veneranta *et al.*, 2013a).

Both ecotypes mature at an age of 3–7 years (Lehtonen, 1981). Some females may spawn only every second year. The eggs are fertilized externally and are left to develop at the bottom. Spawners may spend winter in rivers or near estuaries along the coast. Eggs of the migratory ecotype hatch in early spring, at ice breakup or during spring floods. Larvae swim downstream to estuarine bays or the sea, and juveniles migrate to the sea in summer (Larsson *et al.*, 2013). Larvae of the sea-spawning ecotype use the most shallow shore areas as feeding grounds (Hudd *et al.*, 1988; Veneranta *et al.*, 2013b).

4.11.3 Stock structure and population dynamics

No population estimates are available for coastal whitefish populations. Catches of both sea-spawning and river-spawning ecotypes have declined drastically during recent decades in the coastal areas of the Baltic Sea (OSF, 2015; Havs och Vattenmyndigheten, 2016). In the Finnish coastal area, most of the migratory whitefish caught are 4–5 years of age, and are caught just before or after first spawning (Leskelä *et al.*, 2004). Fishing pressure on the sea-spawning ecotype is lower, and has decreased in the Gulf of Bothnia during the last decade. Age–size data available from the Finnish coast indicates less fishing pressure (Natural Resources Institute, unpub. data). However, data from monitoring fishing in the southern Bothnian Sea in Sweden suggests that the proportion of older fish (> five years) has decreased since the early 1980s (Havs och Vattenmyndigheten, 2016). The increase in seal stocks may also have affected the whitefish stocks and catches in the Baltic Sea area (Lundström *et al.*, 2010). Seals will consume whitefish, decreasing their numbers, and they will disturb gillnet fishing, thus decreasing catches.

4.11.4 Genetics

The whitefish is a species with unclear taxonomy, consisting of a group of populations, forms, or species. The systematics of the species are acknowledged to be complex and confusing, with the occurrence of numerous ecological forms. Members of the genus *Coregonus* readily hybridize with other *Coregonus* species and populations (Østbye *et al.*, 2005). Based on genetic studies, the ecotypes belong to the same species and reflect ecological divergence since the last glaciation. The Baltic populations are structured, have medium-to-high differentiation, high population diversity, and mainly unknown effective population sizes (McCairns *et al.*, 2012). Temporal data are available for some hatchery populations. Genetic risks include loss of local adaptation and genetic swamping.

4.11.5 International status with country highlights

International

OSPAR Convention:	Not listed
EC Habitats Directive:	Annexes II and IV (2007; misidentified as <i>C. ox-yrinchus</i> , Ref. 88171) Annex V - unfavourable
CITES:	Not listed
Bern Convention:	Appendix III (2002)
Bonn Convention:	Not listed
IUCN Criteria:	A2bd
Global IUCN Red List Category:	VU
European IUCN Red List Category:	VU
HELCOM Red List:	EN

Past and current threats (Habitats Directive Article 17 codes): Migration barriers (J03.02.01), fishing (both commercial and recreational) (F02), bycatch (F02).

Future threats (Habitats Directive Article 17 codes): Migration barriers (J03.02.01), fishing (both commercial and recreational) (F02), bycatch (F02).

National IUCN Red List classifications (HELCOM Red List)

Denmark:	– / LC
Estonia:	Fisheries regulations / DD
Finland:	Stocking of specimens; fishing is not allowed in rivers during spawning time; mesh size and fishing effort restrictions for gill-nets in the sea area / EN (river spawning) and VU (sea spawning)
Germany:	Stocking of specimens / NT (Baltic Sea)
Latvia:	Special status by Council of Ministers regulations / VU (river spawning)
Lithuania:	Minimum landing size (36 cm) / <i>C. lavaretus holas</i> , (I) Indeterminate
Poland:	Stocking of specimens; minimum landing size (40 cm); protection period (1 October–31 December); mesh size restrictions / DD
Russian Federation:	– / EN
Sweden:	Protected from fishing during spawning time (1 November–15 December in the county of Gotland, and 15 October–30 November in the county of Gävleborg); a no-take area in the southern Bothnian Sea was implemented between 2011 and 2016 / LC

4.11.6 Distribution summary

European distribution (Figure 4.12)

- In coastal areas: Finland, Sweden, Estonia, Denmark, Poland, Latvia, Lithuania, and the Russian Federation.
- Baltic Sea coastal areas - southeast North Sea Basin: Ems, Weser, and Elbe drainages, and small rivers of Schleswig-Holstein and Denmark.
- Landlocked in several lakes in France, Switzerland, Austria, Germany, Finland, Sweden, Estonia, Norway, Slovakia, and Russian Federation.

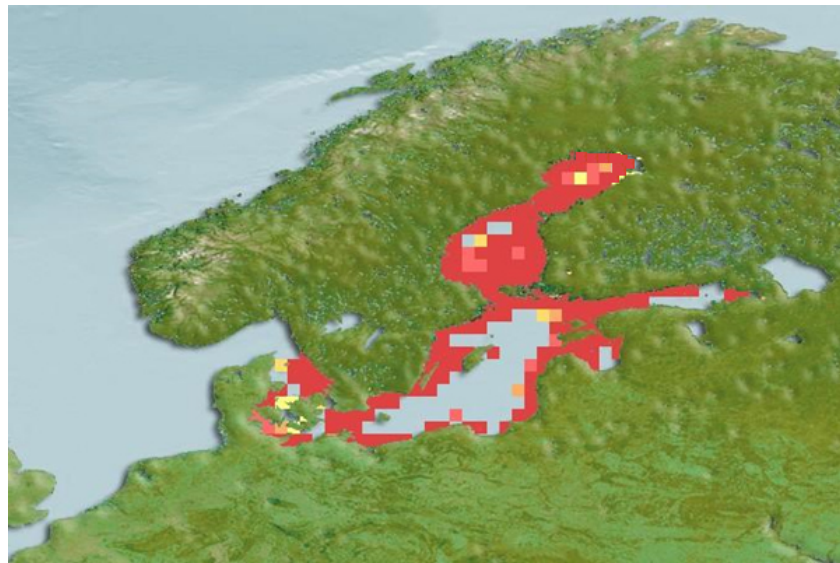


Figure 4.12. Distribution of whitefish (Froese and Pauly, 2015).

4.11.7 Country-by-country changes in distribution

Introduced and transplanted in many drainages within its native range and outside its range, westward to the Rhine drainage. The distribution range of whitefish has remained stable during recent decades, although fishing monitoring in the Baltic Sea (Estonia, Finland, Lithuania and Sweden) indicates remarkable decreases in catches during recent decades for both ecotypes (Verliin *et al.*, 2013; Havs och Vattenmyndigheten, 2016). A no-take zone in the southern Bothnian Sea resulted in an increased stock in this area (Florin *et al.*, 2016). In the northern Baltic Sea, mainly in Finland, the whitefish fishery are supported by vast stockings of larvae and one-summer-old fingerlings. Without stocking, the populations targeted by fishing would most likely rapidly decline (Leskelä *et al.*, 2004; Jokikokko and Huhmarniemi, 2014).

4.11.8 Threats

Whitefish is threatened by a variety of factors affecting reproduction in rivers and coastal areas. Dams and weirs in rivers hamper spawning migrations, and degraded water quality affects reproduction possibilities. Eutrophication of the reproduction habitats, especially in coastal areas, and climate change, have had an impact on the sea-spawning ecotype, especially in southern areas. Being a target species for fishery and a bycatch of other fishery also affects population structure and reproduction capacity. Introduction of *Coregonus* species and populations from other areas has mixed the original stocks because *Coregonus* specimens readily hybridize.

4.11.9 Conclusion – future

Management recommendations:

- To maintain sustainable whitefish fishery and stocks, fishing regulations, especially for gillnets, evaluated successful stockings, and habitat restorations should be continued and used.
- In restocking, local broodstocks should be used.
- Where whitefish have been switched for other salmonids in compensatory stockings, this should be reversed.
- To enhance the natural reproduction of anadromous whitefish, migration barriers should be removed, suitable fish ways constructed, and water quality should be improved.
- In coastal areas, the main reason for decreasing stocks of sea-spawning whitefish are eutrophication and iceless winters. To protect the remaining stocks, local fishing regulations are suggested as well as spawning habitat restoration and protection.

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1. Status of data-limited species

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2.1. Allis shad/Twaite shad

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2.2. European river lamprey

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2.3. Sea lamprey

Pedro R. Almeida, Catarina S. Mateus, Ann-Britt Florin, Bernardo Quintella, Carlos Antunes, Fernando Cobo, Françoise Daverat, Jorge Bochechas, James King, Sandra Barca Bravo, and Sergio Silva Bautista

2.4. Sea trout

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2.5.1. Shortnose sturgeon

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2.5.2. Adriatic sturgeon

Jörn Gessner, Marie-Laure Acolas, and Paolo Bronzi

2.5.3. Baltic sturgeon

Jörn Gessner and Marie-Laure Acolas

2.5.4. European sturgeon

Marie-Laure Acolas, Geraldine Lassalle, and Jörn Gessner

3.1. European eel

Alan M. Walker

3.2. European flounder

Ester Dias, Françoise Daverat, and Pedro Morais

3.3. Thin-lipped grey mullet

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4.1. Burbot

Mehis Rohtla

4.2. Coastal grayling

Lari Veneranta, Alfred Sandström, and Erkki Jokikokko

4.3. Ide

Mehis Rohtla

4.4. Perch

Mehis Rohtla

4.5. Pike

Mehis Rohtla

4.6. Roach

Mehis Rohtla

4.7. Smelt

Jan Breine, Andy Moore, Mervyn Roos, Jan Krankenburg, Karen van de Wolfshaar, Mehis Rohtla, Janis Birzaks, Ann-Britt Florin, Ian McCarthy¹, Russell Poole, and James J. King

4.8. Three-spined and nine-spined stickleback

Ann-Britt Florin, Pedro Almeida, Carlos Alexandre, Jan Breine, Fernando Cobo, Mehis Rohtla, and Jens Olsson

4.9. Vendace

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4.10. Vimba bream

Mehis Rohtla

4.11. Whitefish

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8 ANNEX 1. *Alosa* spp. - Current and historical distribution

Table A1.1. Current best estimates as to which rivers support self-sustaining anadromous populations of *Alosa alosa* and *Alosa fallax* spp. Parentheses indicate that shad have been recorded in marked rivers but there is uncertainty as to whether a self-sustaining population exists.

Atlantic, North Sea, and Baltic populations				
Country	River Basin	<i>A. alosa</i>	<i>A. fallax</i> spp.	Reference/Comment
Norway				
Sweden				
Finland				
Russian Federation				
Estonia				
Latvia				
Lithuania	Nemunas		✓	Švagždys (1999)
Germany	Elbe		✓	Hass (1965, 1968, 1969); Thiel <i>et al.</i> (1996)
	Ems (Eems/Dollard)		✓	Hadderingh and Jager (2002); Kleef and Jager (2002)
	Rhine	✓	✓	Hundt <i>et al.</i> (2015)
Denmark				
Netherlands	Westerschelde		✓	Jager and Kranenbarg (2004)
Belgium	Scheldt (Zeeschelde)		✓	Maes <i>et al.</i> (2005, 2008); Breine <i>et al.</i> (2011a, 2011b)
UK	Severn		✓	Aprahamian and Aprahamian (1990)
	Wye		✓	Aprahamian and Aprahamian (1990)
	Usk		✓	Aprahamian and Aprahamian (1990)
	Tywi		✓	Aprahamian and Aprahamian (1990)
	Tamar	✓		Hillman (2003)
Ireland	Barrow		✓	King and Roche (2008)
	Suir		(✓)	
	Nore		(✓)	
	Slaney	(✓)	(✓)	
	Munster Blackwater	(✓)	(✓)	

Table A1.1 (continued)	River Basin	<i>A. alosa</i>	<i>A. fallax</i> spp.	Reference/Comment
France (Atlantic)	Orne		✓	Baglinière <i>et al.</i> (2003); Rochard <i>et al.</i> (2007)
	Vire		✓	Baglinière <i>et al.</i> (2003); Rochard <i>et al.</i> (2007)
	Aulne		✓	Véron (1999)
	Blavet		✓	Véron (1999)
	Vilaine	✓	✓	Mennesson-Boisneau and Boisneau (1990); Taverny (1991a, 1991b); Prouzet <i>et al.</i> (1994); Véron (1999); Baglinière (2000)
	Loire	✓	✓	
	Charente	✓	✓	
	Garonne and Dordogne	✓	✓	
	Adour	✓	✓	
	Nivelle	✓	✓	
Spain (Atlantic)	Bidasoa	✓		Leunda <i>et al.</i> (2012)
	Asón	✓		Doadrio <i>et al.</i> (2011)
	Ulla		✓	Cobo <i>et al.</i> (2010); MIGRANET (2012); Bao <i>et al.</i> (2015a, 2015b); Nachón <i>et al.</i> (2013, 2015a)
Portugal and Spain	Minho	✓	✓	Eiras (1980); Assis (1990); Alexandrino (1996a, 1996b); Bao <i>et al.</i> (2015a, 2015b); Costa <i>et al.</i> (2001); Esteves and Andrade (2008); Faria <i>et al.</i> (2012); Mota and Antunes (2011, 2012); MIGRANET (2012); Mota <i>et al.</i> (2015); Nachón <i>et al.</i> (2015a, 2015b)
Portugal	Lima	✓	✓	
	Vouga	✓	✓	
	Mondego	✓	✓	
	Tejo	✓	✓	
	Sado		(✓)	
	Mira		✓	
	Guadiana	✓	✓	Collares-Pereira <i>et al.</i> (2000); Costa <i>et al.</i> (2001); Eiras (1980); Faria <i>et al.</i> (2012)
Morocco (Atlantic)	Sebou		✓	Sabatié (1993); Sabatié and Baglinière (2001)
	Loukos		✓	

Table A1.1 (continued)	River Basin	<i>A. alosa</i>	<i>A. fallax</i> spp.	Reference/Comment
Mediterranean populations				
France (Mediterranean)	Aude		✓	Rameye <i>et al.</i> (1976); Douchement (1981); Changeux and Pont (1995); Le Corre <i>et al.</i> (1997)
	Hérault		✓	
	Vidourle		✓	
	Rhône		✓	
	Argens		✓	
Spain (Mediterranean)	Ebro		✓	López <i>et al.</i> (2007, 2011); Sotelo <i>et al.</i> (2014)
Morocco (Mediterranean)	Oued Moulouya		✓	Sabatié (1993)
Algeria	Chelif		✓	Dieuzeide <i>et al.</i> (1959); Quignard and Kartas (1977)
	Mazafran		✓	
	Seybouse		✓	
	Oubeira		✓	
Tunisia	Miliane		✓	Quignard and Kartas (1977); Kartas (1981)
	Medjerdah		✓	
Libya				
Egypt				
Israel				
Lebanon				
Turkey	Yeşilirmak (Black Sea)		✓	Turan and Basusta (2001)
	Sakarya (Black Sea)		✓	
	Menderes (Aegean Sea)		✓	
	Seyhan		✓	
	Ceyhan		✓	

Table A1.1 (continued)	River Basin	<i>A.alosa</i>	<i>A. fallax spp.</i>	Reference/Comment
Greece	Evros		✓	Barbieri <i>et al.</i> (2015); Bobori <i>et al.</i> (2001); Economidis (1974)
	Filiouris		✓	
	Vistonis basin		✓	
	Nestos		✓	
	Strymon		✓	
	Pinios		✓	
	Acheloos		✓	
	Louros		✓	
	Arachthos		✓	
	Kalamas		✓	
	Thyamis		✓	
	Loudias		(✓)	
	Aliakmon		(✓)	
	Axios		(✓)	
Albania				
Montenegro				
Albania/Montenegro	Bojana		✓	Vukovic (1961b); Vukovic and Ivanovic (1971); Kosoric <i>et al.</i> , (1989a, 1989b); Rakaj and Crivelli (2001)
Croatia/Bosnia and Herzegovina	Neretva		✓	Vukovic (1961b); Vukovic and Ivanovic (1971); Kosoric <i>et al.</i> , (1989a, 1989b); Bianco (2002)
Croatia				
Slovenia				
Italy	Piave		✓	Chiesa <i>et al.</i> (2014)
	Brenta		✓	Rizzotti and Gioppato (1997)
	Ombrone		✓	Chiesa <i>et al.</i> (2014); S. Sabatino (pers. comm.)
	Po		✓	Gandolfi and Le Moli (1977); Vitali and Braghieri (1981); Vitali <i>et al.</i> (1983); Chiaudani and Marchetti (1984); Serventi <i>et al.</i> (1990); Bianco (2002); Chiesa <i>et al.</i> (2014)

Table A1.1 (continued)	River Basin	<i>A. alosa</i>	<i>A. fallax</i> spp.	Reference/Comment
Sardinia	Barca		✓	Cottiglia (1968)
	Cedrino		✓	
	Coghinas		✓	
	Flumendosa		✓	
	Posada		✓	
	Temo		✓	
	Tirso		✓	
Corsica	Golo		✓	Casabianca <i>et al.</i> (1972–1973); Kiener (1985); Changeux and Pont (1995); Le Corre <i>et al.</i> (2000); Rougemont (2012)
	Tavignano		✓	

Table A1.2. Rivers where populations of *Alosa alosa* and *Alosa fallax* spp. have become extinct. Parentheses indicate that shad were recorded in marked rivers but there is uncertainty as to whether a self-sustaining population existed.

Country	River basin	<i>A. alosa</i>	<i>A. fallax</i> spp.	Cause	Reference
Atlantic, North Sea, and Baltic populations					
Norway					
Sweden					
Finland					
Russian Federation					
Estonia	Narva		(✓)		Mikelsaar (1984); Saat (2002)
Latvia					
Lithuania					
Poland	Oder		✓		Pęczalska (1973)
	Vistula		✓		Backiel (1995)
Germany	Elbe	(✓)			Wilkens and Köhler (1977); Möller (1989)
	Weser	✓		Overfishing, channelization, and the construction of dams	Busch <i>et al.</i> (1988, 1989)
	Rhine	✓		Overfishing, barriers, and destruction of spawning habitat	de Groot (1989, 1990); Raat (2001)
Denmark					
Netherlands					
Belgium	Meuse	✓	✓	Overexploitation, pollution, habitat destruction, and the building of weirs	Poll (1947); Philippart and Vranken (1981, 1982); Philippart <i>et al.</i> (1988)
	Scheldt	✓			De Selys-Llongchamps (1842); Poll (1945)
UK	Severn	✓		Navigation weirs	Day (1890)
	Trent	✓	✓		Anon. (1622, 1890); Deering (1751)
	Thames		✓	Pollution	Aprahamian and Aprahamian (1990)

Table A1.2 (continued)	River basin	<i>A. alosa</i>	<i>A. fallax</i> spp.	Cause	Reference
Ireland					
France	Seine	✓	✓	Barriers and pollution	Vincent (1889); Roule (1920); Le Clerc (1941); Belliard (1994)
Spain					
Portugal	Douro ¹	✓	✓	Dam construction	Baldaque da Silva (1891); Nobre (1931); Alexandrino (1996b); Costa <i>et al.</i> (2001); Cabral <i>et al.</i> (2005)
	Ave	✓	✓		
	Sado	✓		Possibly pollution	
Morocco	Sebou	✓		Barriers, pollution	Sabatié (1993)
	Bou Regreg	✓			
	Oum er Rbia	✓			
	Mass	✓			
Mediterranean populations					
France	Rhône	(✓)			Le Roux (1928) ; Rameye <i>et al.</i> (1976),
Spain	Ebro	✓		Dam construction	Lozano Rey (1935, 1950); Sostoa and Sostoa (1979); Sostoa and Lobon-Cervia (1989)
Morocco	Oued Moulouya	✓	✓	Dam construction	Sabatié (1993)
Algeria					
Tunisia					
Libya					
Egypt	Nile		✓	Dam construction	Boulenger (1907); Ladiges and Vogt (1965); Bishai and Khalil (1997)
Israel					
Lebanon					
Turkey					

Table A1.2 (continued)	River basin	<i>A. alosa</i>	<i>A. fallax</i> spp.	Cause	Reference
Greece	Nestos		✓	Dam construction	Bobori <i>et al.</i> (2001)
Albania					
Montenegro					
Bosnia and Herzegovina					
Croatia					
Slovenia					
Italy	Arno		✓		Barbieri (1907); D'Ancona (1928); Berg (1933); Ferrero (1951); Bini (1970); Sam-micheli (1998)
	Tiber		✓	Dam construction	
	Volturno		✓		
	Sele		✓		
Sardinia					
Corsica					

¹In the Douro, there are occasional records, but the population are not viable on account of the several dams that exist in this basin (Cabral *et al.*, 2005).

9 ANNEX 2. *Alosa* spp. - Monitoring

The information presented here was gathered at the catchment scale and is a summary of the data collected by the ICES Lamprey and Shad Working Group (Lambert *et al.*, 2015). It therefore, does not claim to be exhaustive, but rather it gives a first impression of what stocks exist in Europe. It needs to be expanded, especially with information from northern European and Baltic countries and from the Mediterranean.

The countries identified three main aims for their monitoring programme: conservation, fishery and conservation, and fishery. The methodologies used consisted both of fishery-dependent and fishery-independent methods. For fishery-dependent methods, there were three main sources: official catch statistics, unofficial catch statistics, and mark-recapture. For fishery-independent methods, assessments were made using fish pass surveys, bottom sampler egg traps, bongo netting for post-larval shads (horizontal-haul zooplankton net), anchor net, fykenet and seinenet surveys, spawning event surveys, and power station surveys. Surveys targeted all life stages and environments (adults in rivers, juveniles in rivers/estuaries/sea, adults at sea). The main sources of funding identified were: local or regional funding, national funding, European funding [INTERREG, SUDOE, SAC, water framework directive (WFD), data collection framework (DCF)], and research and development programmes from private or public institutes.

No information was found as to whether the monitoring programmes were suitable to meet their objectives.

Table A2.1. Monitoring programmes and data for *Alosa* spp.

Country	Monitoring and data
Belgium	Adult and juvenile surveys with anchor nets and fykenets, power station surveys (Maes <i>et al.</i> , 2008)
Finland	Commercial fishery catch statistics, data collection in EU-data frame collection and occasional projects
France	Official catch statistics, fish pass surveys; spawning event surveys, beach-seine surveys (Boisneau <i>et al.</i> , 2008); juvenile surveys (Girardin and Castelnaud, 2013)
Germany	Fish pass surveys; juvenile surveys (Hundt <i>et al.</i> , 2015)
Ireland	Juvenile surveys in estuaries (King and Roche, 2008); official catch statistics
Spain	Fish pass surveys, adult and juvenile experimental fishing (Nachón, 2016)
Portugal	Fish pass surveys (Almeida <i>et al.</i> , 2015), adult and juvenile experimental fishing (Mota, 2014), Official catch statistics, surveys of professional fisher (Almeida <i>et al.</i> , 2015).
UK	Fish pass surveys; egg surveys (Caswell and Aprahamian, 2001) and power station samples (Holmes and Henderson, 1990)

10 ANNEX 3. Sea trout – Additional figures and tables

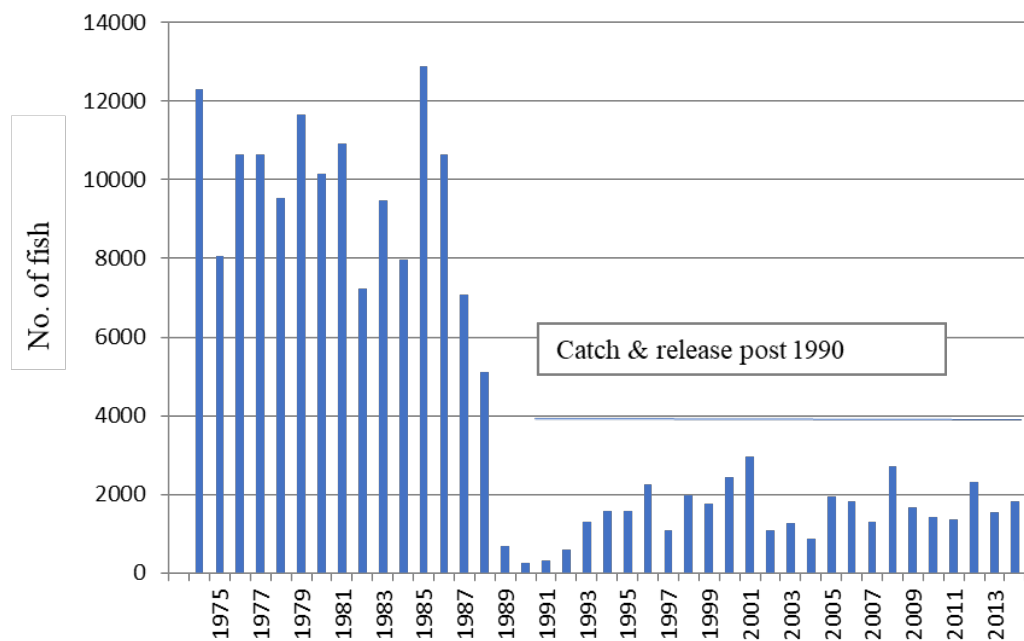


Figure A3.1. Sea trout rod catch for eighteen Connemara (Ireland) fishery (1974–2014).

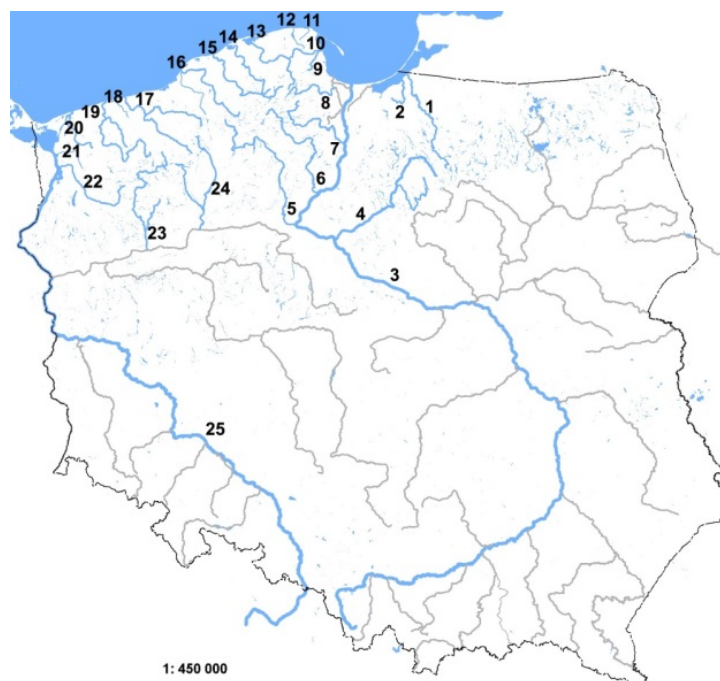


Figure A3.2. Polish sea trout rivers.

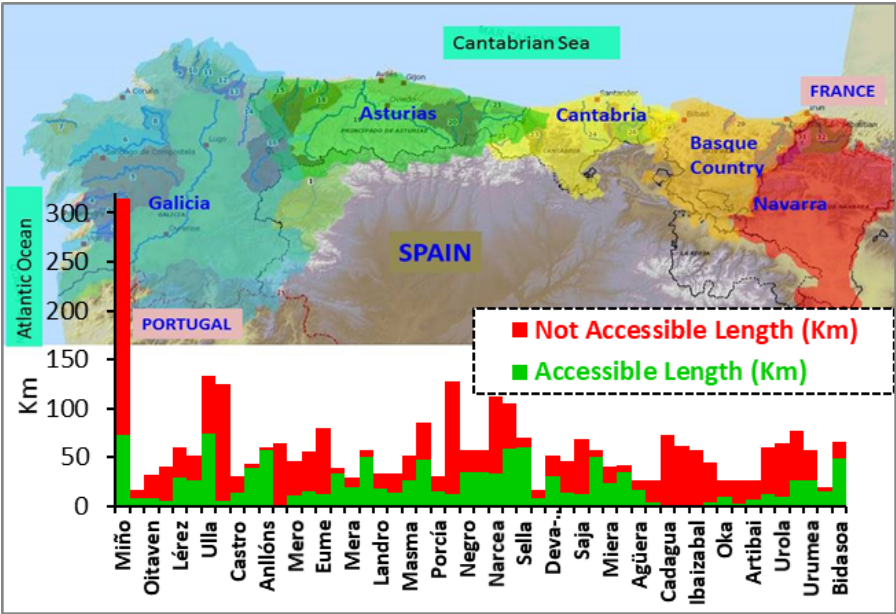


Figure A3.3. Administrative division in Spanish north and northwest areas and accessible/not accessible length for sea trout in 49 Spanish main rivers draining to the Atlantic Ocean and Cantabrian Sea.

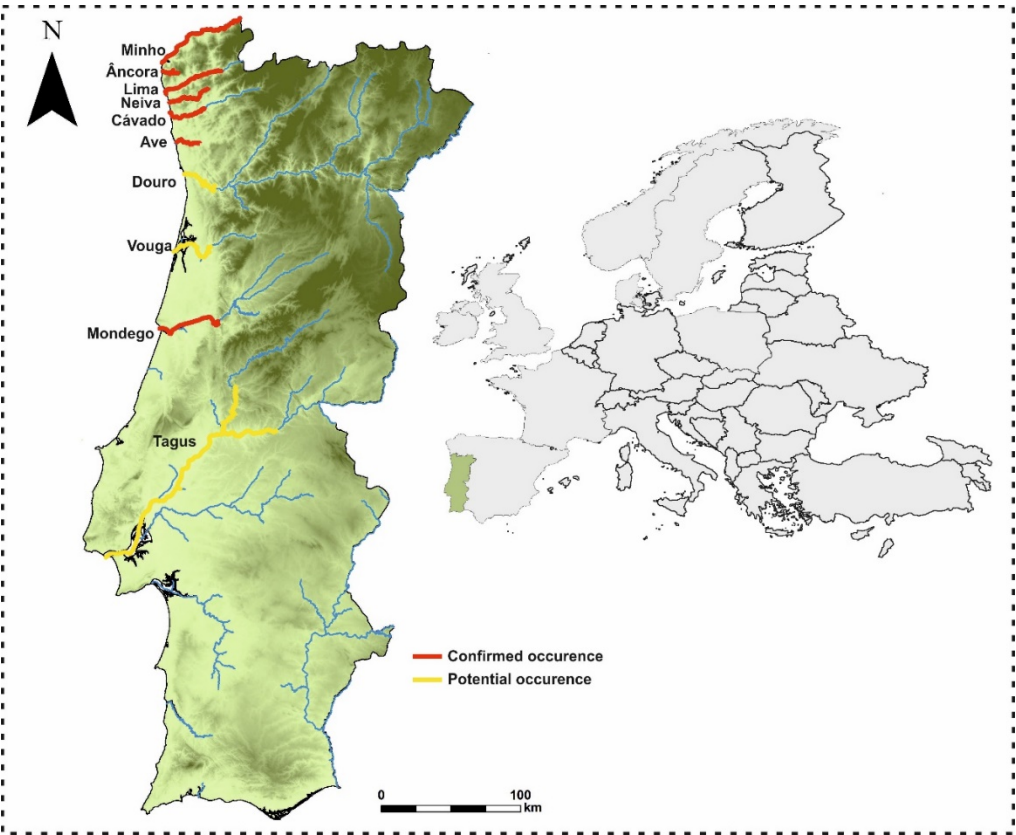


Figure A3.4. Sea trout distribution in Portugal. Upstream limits are defined by the first unsurmountable barriers.

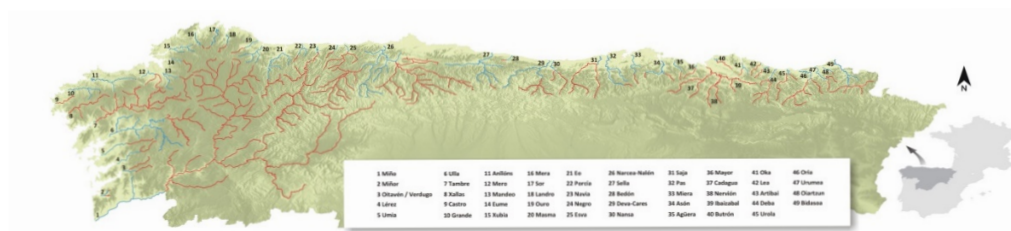


Figure A3.5. Map showing sea trout distribution in Spain, with the accessible (blue) and not accessible (red) river length in the 49 Spanish major rivers that flow into the Atlantic Ocean and into the Cantabrian Sea.

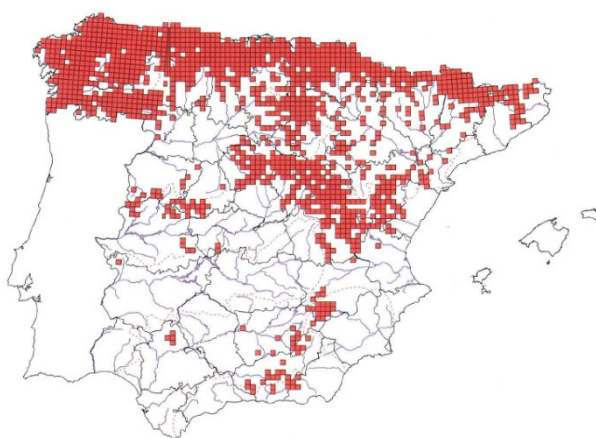


Figure A3.6. *Salmo trutta* distribution in Spain, Atlas and Red Book of the Freshwater Fish from Spain (Doadrio, 2001).

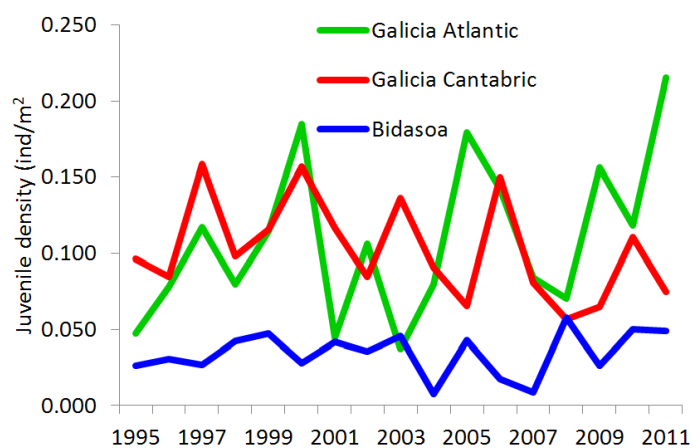


Figure A3.7. Electrofishing surveys of sea trout juveniles in Spain.

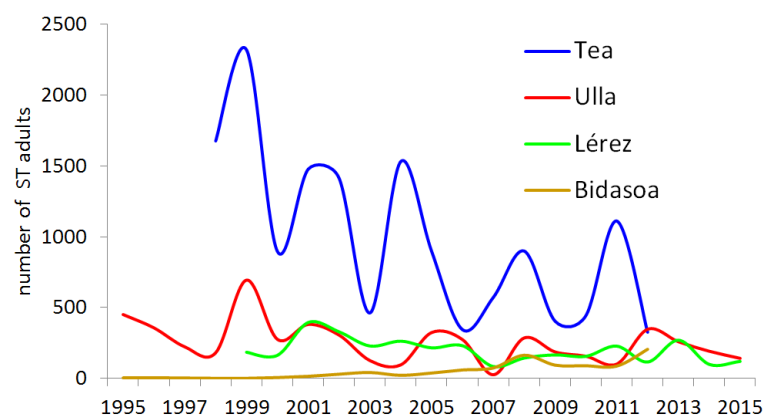


Figure A3.8. Number of sea trout adults catch at 4 regular traps in Spanish rivers (Bidasoa River-Bera, Lérez River-Bora, Ulla River-Ximonde, and Tea River-A Freixa).

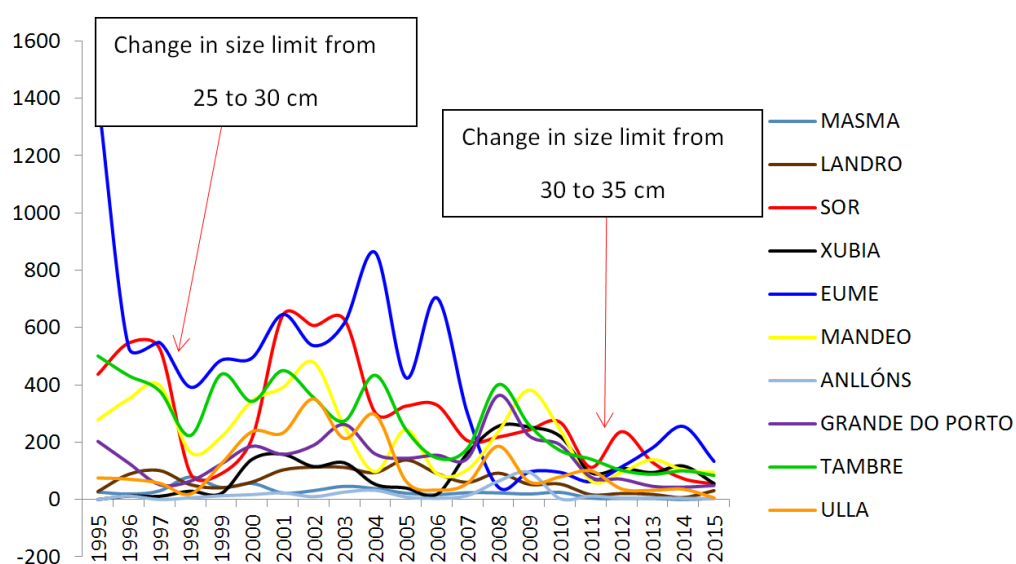


Figure A3.9. Galicia (Spain) Sea trout recreational fishery: official catch in 10 rivers (1995–2015).

Table A3.1. Systems in Ireland with sea trout populations.

River basin district	Fishery district	Fisheries system	Latitude (N)	Longitude (W)
ERBD	Drogheda	Boyne (River)	53°43'8.927"	6°25'36.414"
ERBD	Drogheda	Delvin	53°37'59.688"	6°12'58.181"
ERBD	Drogheda	Nanny (River)	53°40'11.794"	6°16'43.079"
ERBD	Dublin	Ballough (Corduff)	53°29'59.602"	6°11'7.227"
ERBD	Dublin	Ballybohil (Ballyboughal)	53°29'55.671"	6°11'8.264"
ERBD	Dublin	Broad Meadow Water	53°28'8.066"	6°12'33.368"
ERBD	Dublin	Dargle (River)	53°12'22.182"	6°6'30.056"
ERBD	Dublin	Dodder (River)	53°20'37.448"	6°13'46.226"
ERBD	Dublin	Liffey (River)	53°20'46.878"	6°18'55.547"
ERBD	Dublin	Newcastle	53°4'40.993"	6°2'13.726"
ERBD	Dublin	Newtownmountkennedy	53°5'31.671"	6°2'36.194"
ERBD	Dublin	Rathnew (River)	53°0'11.203"	6°3'22.613"
ERBD	Dublin	Shanganagh	53°14'46.078"	6°6'48.193"
ERBD	Dublin	Tolka (River)	53°21'55.568"	6°14'55.946"
ERBD	Dublin	Vartry (River)	53°0'24.677"	6°3'21.579"
ERBD	Dublin	Ward (River)	53°28'12.238"	6°12'46.259"
ERBD	Dundalk	Ryland (River)	54°4'42.520"	6°14'47.502"
ERBD	Wexford	Avoca (River)	52°48'4.249"	6°9'52.111"
ERBD	Wexford	Potters (River)	52°53'45.335"	6°2'57.423"
ERBD	Wexford	Redcross (River)	52°50'32.844"	6°6'5.701"
ERBD	Wexford	Three Mile (River)	52°55'18.813"	6°1'55.245"
NBRBD	Drogheda	Termonfeckin (Stream)	53°45'32.855"	6°14'56.164"
NBRBD	Dundalk	Ballymascanlan (Flurry) (River)	54°1'28.281"	6°21'4.854"
NBRBD	Dundalk	Castletown (Piedmont) (Cooley) (River)	53°59'6.625"	6°13'19.158"
NBRBD	Dundalk	Castletown (River)	54°1'11.950"	6°25'8.234"
NBRBD	Dundalk	Dee (River)	53°52'19.186"	6°21'18.438"
NBRBD	Dundalk	Fane (River)	53°56'33.782"	6°24'7.982"
NBRBD	Dundalk	Glyde (River)	53°53'16.551"	6°21'45.424"
NWRBD	Ballyshannon	Abbey (River)	54°30'27.186"	8°12'14.277"
NWRBD	Ballyshannon	Ballintra (River)	54°36'7.756"	8°8'51.653"
NWRBD	Ballyshannon	Bradoge (River)	54°28'41.816"	8°16'52.041"
NWRBD	Ballyshannon	Bungosteen (River)	54°39'2.683"	8°25'11.718"
NWRBD	Ballyshannon	Drowes (River)	54°28'24.477"	8°19'16.796"
NWRBD	Ballyshannon	Duff (River)	54°27'51.384"	8°22'51.996"

Table A3.1 (continued)				
River basin district	Fishery district	Fisheries system	Latitude (N)	Longitude (W)
NWRBD	Ballyshannon	Durnesh	54°34'12.789"	8°10'51.464"
NWRBD	Ballyshannon	Eany (Water)	54°39'32.461"	8°16'21.375"
NWRBD	Ballyshannon	Erne (River)	54°29'59.892"	8°10'30.236"
NWRBD	Ballyshannon	Eske (River)	54°39'17.434"	8°6'41.544"
NWRBD	Ballyshannon	Glen (River)	54°39'0.520"	8°38'34.892"
NWRBD	Ballyshannon	Glenaddragh	54°37'37.269"	8°36'11.837"
NWRBD	Ballyshannon	Laghy (Stream)	54°37'6.146"	8°6'32.549"
NWRBD	Ballyshannon	Oily (River)	54°38'6.108"	8°23'49.297"
NWRBD	Letterkenny	Abberachrin (Duvoge) (River)	54°48'3.976"	8°27'41.792"
NWRBD	Letterkenny	Aghaweel	55°11'10.528"	7°31'19.875"
NWRBD	Letterkenny	Ballyboe (River)	55°17'33.447"	7°14'56.171"
NWRBD	Letterkenny	Big (Burn)	55°8'34.419"	7°41'35.460"
NWRBD	Letterkenny	Bracky (River)	54°45'25.173"	8°25'52.796"
NWRBD	Letterkenny	Bredagh (River)	55°11'14.865"	7°2'47.935"
NWRBD	Letterkenny	Bunlin (River)	55°5'48.358"	7°43'3.631"
NWRBD	Letterkenny	Burnfoot	55°3'20.002"	7°26'47.043"
NWRBD	Letterkenny	Burnside (River)	55°8'30.057"	7°40'10.081"
NWRBD	Letterkenny	Clady (River)	55°3'28.761"	8°18'20.712"
NWRBD	Letterkenny	Clonmany (River)	55°16'41.639"	7°26'13.742"
NWRBD	Letterkenny	Crana (River)	55°8'22.575"	7°27'43.649"
NWRBD	Letterkenny	Culdaff (River)	55°17'16.161"	7°10'4.183"
NWRBD	Letterkenny	Culoort (Keenagh) (River)	55°21'34.567"	7°21'0.452"
NWRBD	Letterkenny	Derryart (Duvoge) (River)	55°9'19.489"	7°54'52.495"
NWRBD	Letterkenny	Donagh (River)	55°16'40.959"	7°15'58.908"
NWRBD	Letterkenny	Duogh (Owenwee?) (River)	54°45'40.878"	8°31'17.313"
NWRBD	Letterkenny	Dungloe (Dunglow) (River)	54°56'57.772"	8°21'52.031"
NWRBD	Letterkenny	Duntally River	55°7'49.642"	7°52'8.967"
NWRBD	Letterkenny	Faymore (River)	55°8'0.280"	7°54'28.284"
NWRBD	Letterkenny	Glenalla	55°4'27.318"	7°35'56.561"
NWRBD	Letterkenny	Glenna (River)	55°7'10.067"	8°8'20.723"
NWRBD	Letterkenny	Glennagannon (Loughinn) (River)	55°16'42.853"	7°15'8.797"
NWRBD	Letterkenny	Glenvar (River)	55°10'14.050"	7°34'35.907"
NWRBD	Letterkenny	Gweebarra (River)	54°54'20.490"	8°12'22.603"

Table A3.1 (continued)				
River basin district	Fishery district	Fisheries system	Latitude (N)	Longitude (W)
NWRBD	Letterkenny	Gweedore (Crolly) (River)	55°2'14.991"	8°16'21.909"
NWRBD	Letterkenny	Isle (Burn)	54°57'58.059"	7°37'22.397"
NWRBD	Letterkenny	Lackagh (River)	55°7'27.090"	7°51'1.215"
NWRBD	Letterkenny	Leannan (River)	55°2'14.353"	7°38'55.593"
NWRBD	Letterkenny	Loughkeel (Burn)	55°5'55.142"	7°42'47.117"
NWRBD	Letterkenny	Mill (River)	55°7'45.805"	7°27'28.286"
NWRBD	Letterkenny	Owenawilin (River)	55°7'37.179"	8°9'53.308"
NWRBD	Letterkenny	Owenea (River)	54°46'28.874"	8°24'40.190"
NWRBD	Letterkenny	Owennamarve (River)	54°54'23.504"	8°20'28.420"
NWRBD	Letterkenny	Owentocker (River)	54°45'49.730"	8°25'12.623"
NWRBD	Letterkenny	Owentully	55°7'30.656"	8°9'22.132"
NWRBD	Letterkenny	Owenwee (Yellow) (River)	54°39'0.635"	8°38'37.627"
NWRBD	Letterkenny	Ray (River)	55°9'35.281"	8°4'54.290"
NWRBD	Letterkenny	Straid (Straths) (River)	55°17'3.350"	7°19'34.524"
NWRBD	Letterkenny	Swilly (River)	54°56'37.461"	7°45'0.754"
NWRBD	Letterkenny	Tullaghobegly (River)	55°8'10.217"	8°7'15.511"
SERBD	Lismore	Goish	52°4'38.964"	7°51'5.923"
SERBD	Waterford	Annestown (Stream)	52°8'27.045"	7°16'27.326"
SERBD	Waterford	Ballymoat	52°14'31.691"	7°12'39.979"
SERBD	Waterford	Barrow (River)	52°30'8.155"	6°56'23.383"
SERBD	Waterford	Black Water	52°16'38.452"	7°8'30.042"
SERBD	Waterford	Brickey (River)	52°3'51.321"	7°38'47.780"
SERBD	Waterford	Clodiagh (River)	52°17'7.253"	7°18'21.805"
SERBD	Waterford	Colligan (River)	52°6'21.784"	7°38'44.070"
SERBD	Waterford	Corock (River)	52°16'19.735"	6°44'28.358"
SERBD	Waterford	Daligan (Dalligan) River	52°6'25.612"	7°30'38.862"
SERBD	Waterford	Dawn	52°15'26.138"	7°15'34.415"
SERBD	Waterford	John's River (River)	52°15'28.942"	7°6'17.207"
SERBD	Waterford	Lingaun (River)	52°20'41.224"	7°22'23.773"
SERBD	Waterford	Mahon (River)	52°8'35.997"	7°22'20.713"
SERBD	Waterford	Nore (River)	52°29'42.954"	7°3'44.094"
SERBD	Waterford	Owenduff (River)	52°16'20.006"	6°47'22.761"
SERBD	Waterford	Pil	52°20'1.931"	7°20'4.484"
SERBD	Waterford	Pollmounty (River)	52°27'59.104"	6°54'33.232"
SERBD	Waterford	Suir (River)	52°20'59.220"	7°26'44.340"
SERBD	Waterford	Tay (River)	52°7'29.247"	7°27'32.932"
SERBD	Waterford	Whelanbridge (River)	52°14'45.533"	7°13'57.793"
SERBD	Wexford	Blackwater (River)	52°25'47.111"	6°19'49.623"
SERBD	Wexford	Duncormick	52°13'41.077"	6°39'23.621"
SERBD	Wexford	Kilgorman (Inch) (Clonough) (River)	52°42'43.228"	6°10'16.574"

Table A3.1 (continued)				
River basin district	Fishery district	Fisheries system	Latitude (N)	Longitude (W)
SERBD	Wexford	Owenavorrigh (River)	52°39'5.411"	6°13'42.293"
SERBD	Wexford	Slaney (River)	52°30'8.527"	6°33'57.639"
SERBD	Wexford	Sow (River)	52°22'32.873"	6°27'30.430"
SHRBD	Kerry	Lee (River)	51°53'45.473"	8°31'50.346"
SHRBD	Limerick	Ahacronane (River)	52°35'46.776"	9°3'38.845"
SHRBD	Limerick	Annageeragh (River)	52°46'50.879"	9°28'36.645"
SHRBD	Limerick	Aughaveemagh (Aughaveema) River	52°47'35.294"	9°29'5.527"
SHRBD	Limerick	Aughyvackeen (Dealgh) (Kilshanny) (River)	52°57'42.879"	9°19'5.152"
SHRBD	Limerick	Ballincurra (Creek)	52°38'13.740"	8°38'1.187"
SHRBD	Limerick	Ballyvaskin (River)	52°52'6.337"	9°25'42.374"
SHRBD	Limerick	Brick (River)	52°26'40.383"	9°37'33.145"
SHRBD	Limerick	Cloon (River)	52°38'39.840"	9°16'7.801"
SHRBD	Limerick	Cloonbony (River)	52°51'35.586"	9°25'51.841"
SHRBD	Limerick	Crompaun (River)	52°40'25.364"	8°42'20.569"
SHRBD	Limerick	Deel (River)	52°36'1.285"	8°58'24.279"
SHRBD	Limerick	Doonbeg (River)	52°43'54.180"	9°31'24.838"
SHRBD	Limerick	Feale (River)	52°25'51.769"	9°32'32.557"
SHRBD	Limerick	Fergus (River)	52°48'55.913"	8°57'41.755"
SHRBD	Limerick	Freagh (River)	52°40'32.118"	8°42'16.473"
SHRBD	Limerick	Galey (River)	52°26'41.809"	9°33'39.960"
SHRBD	Limerick	Inagh (River)	52°56'24.176"	9°17'48.166"
SHRBD	Limerick	Kildeema (Bealaclogga) (River)	52°50'13.536"	9°25'55.073"
SHRBD	Limerick	Maigue (River)	52°34'8.192"	8°47'3.075"
SHRBD	Limerick	Moy (River)	54°6'44.929"	9°9'10.625"
SHRBD	Limerick	Owenagarney (Ratty) (Bunratty) (River)	52°42'53.256"	8°46'46.217"
SHRBD	Limerick	Shannon (River)	52°40'52.839"	8°36'34.754"
SHRBD	Limerick	Skivileen (Creagh) (Cree) (River)	52°44'48.157"	9°30'20.453"
SWRBD	Cork	Ardigeen (River)	51°39'45.356"	8°46'38.517"
SWRBD	Cork	Ardrigole (River)	51°41'40.773"	9°43'14.330"
SWRBD	Cork	Bandon (River)	51°45'55.973"	8°40'0.483"
SWRBD	Cork	Bawnaknockane	51°34'7.055"	9°27'27.188"
SWRBD	Cork	Coomhola (River)	51°44'9.572"	9°26'56.876"
SWRBD	Cork	Dungourney (River)	51°54'40.795"	8°10'22.090"
SWRBD	Cork	Four Mile Water (Durrus)	51°37'11.677"	9°31'32.578"
SWRBD	Cork	Glashaboy (River)	51°55'1.821"	8°23'56.958"
SWRBD	Cork	Glengarriff (River)	51°44'57.186"	9°33'3.153"
SWRBD	Cork	Ilen (River)	51°33'37.799"	9°15'46.981"

Table A3.1 (continued)				
River basin district	Fishery district	Fisheries system	Latitude (N)	Longitude (W)
SWRBD	Cork	Leamawaddra River	51°33'7.845"	9°24'50.778"
SWRBD	Cork	Lee (River)	52°15'30.444"	9°43'37.786"
SWRBD	Cork	Mealagh (River)	51°41'40.023"	9°26'22.162"
SWRBD	Cork	Owenboy (Owenabue) (River)	51°48'48.081"	8°23'42.693"
SWRBD	Cork	Owennacurra (River)	51°54'42.074"	8°10'32.662"
SWRBD	Cork	Owvane (River)	51°43'24.149"	9°26'22.132"
SWRBD	Cork	Roury (River)	51°33'29.729"	9°2'42.034"
SWRBD	Cork	Roxboro (Dungourney) (River)	51°54'39.099"	8°10'31.552"
SWRBD	Cork	Stick (River)	51°44'5.387"	8°29'5.706"
SWRBD	Kerry	Behy (River)	52°3'37.872"	9°57'25.230"
SWRBD	Kerry	Blackwater (River)	51°51'20.532"	9°44'37.760"
SWRBD	Kerry	Caragh (River)	52°4'6.705"	9°54'13.539"
SWRBD	Kerry	Carhan (River)	51°56'57.367"	10°12'15.378"
SWRBD	Kerry	Cloonee (River)	51°48'23.364"	9°46'26.540"
SWRBD	Kerry	Coomnahorna	51°45'49.041"	10°6'42.189"
SWRBD	Kerry	Cottoners (River)	52°6'1.454"	9°46'17.572"
SWRBD	Kerry	Croanshagh (Croansaght) (River)	51°45'12.881"	9°46'53.642"
SWRBD	Kerry	Currane (Waterville) (River)	51°49'11.002"	10°10'32.602"
SWRBD	Kerry	Emlagh (River)	52°8'35.437"	9°57'30.747"
SWRBD	Kerry	Emlaghmore (River)	51°50'38.523"	10°14'25.911"
SWRBD	Kerry	Feohanagh (River)	52°12'59.819"	10°21'13.042"
SWRBD	Kerry	Ferta (River)	51°57'51.502"	10°10'52.125"
SWRBD	Kerry	Finnihy (River)	51°52'26.468"	9°35'49.822"
SWRBD	Kerry	Gowla	51°46'37.763"	10°2'18.992"
SWRBD	Kerry	Inny (Knockmoyle) (River)	51°51'29.661"	10°10'22.305"
SWRBD	Kerry	Kealinda (River)	51°41'17.513"	9°58'7.870"
SWRBD	Kerry	Laune (River)	52°6'2.113"	9°45'33.622"
SWRBD	Kerry	Lough Fadda (Stream system)	51°43'1.401"	9°57'28.359"
SWRBD	Kerry	Maine (River)	52°10'48.872"	9°38'25.807"
SWRBD	Kerry	Milltown (River)	52°8'31.418"	10°17'16.988"
SWRBD	Kerry	Owenalongdrig River	52°8'17.312"	10°12'5.890"
SWRBD	Kerry	Owenascaul (River)	52°8'12.078"	10°3'3.634"
SWRBD	Kerry	Owencashla (River)	52°14'22.759"	9°58'25.084"
SWRBD	Kerry	Owenmore (River)	53°26'10.056"	9°52'14.296"
SWRBD	Kerry	Owennafeanna (River)	52°15'50.799"	10°9'44.908"
SWRBD	Kerry	Owenshagh (River)	51°45'55.877"	9°46'30.667"
SWRBD	Kerry	Owreagh (River)	51°50'1.975"	9°54'31.083"

Table A3.1 (continued)				
River basin district	Fishery district	Fisheries system	Latitude (N)	Longitude (W)
SWRBD	Kerry	Roughty (River)	51°53'17.240"	9°32'13.306"
SWRBD	Kerry	Sheen (River)	51°52'27.089"	9°33'46.423"
SWRBD	Kerry	Sneem (River)	51°50'15.942"	9°54'1.848"
SWRBD	Kerry	Staigue	51°46'38.864"	10°2'5.896"
SWRBD	Lismore	Blackwater (River)	52°8'34.069"	7°54'51.360"
SWRBD	Lismore	Bride (River)	52°6'4.971"	7°59'41.484"
SWRBD	Lismore	Finisk (River)	52°7'8.596"	7°49'31.982"
SWRBD	Lismore	Glenshelane (River)	52°8'41.719"	7°51'16.213"
SWRBD	Lismore	Licky (River)	51°59'50.346"	7°49'34.105"
SWRBD	Lismore	Tourig (River)	51°58'31.069"	7°54'3.686"
SWRBD	Lismore	Womanagh (River)	51°55'31.037"	7°57'1.218"
WRBD	Ballina	Ballinglen (River)	54°17'42.569"	9°22'58.288"
WRBD	Ballina	Belderg (River)	54°18'40.038"	9°33'2.600"
WRBD	Ballina	Bellawady (River)	54°12'41.410"	9°5'57.978"
WRBD	Ballina	Brusna (River)	54°7'11.783"	9°8'28.976"
WRBD	Ballina	Cloonaghmore (Palmerstown) (River)	54°13'30.836"	9°16'9.622"
WRBD	Ballina	Cloonalaghan (River)	54°16'14.862"	9°14'50.995"
WRBD	Ballina	Easky (River)	54°17'20.653"	8°57'34.650"
WRBD	Ballina	Finned (Owenykeevan) (River)	54°17'29.331"	9°0'0.751"
WRBD	Ballina	Glencullen (Bellanaminnaun) (River)	54°17'57.479"	9°23'29.257"
WRBD	Ballina	Glenulra	54°18'31.015"	9°26'17.872"
WRBD	Ballina	Leaffony (River)	54°16'2.977"	9°3'43.877"
WRBD	Ballina	Moy (River)	52°55'11.661"	9°21'1.471"
WRBD	Ballinakill	Ballinaboy (River)	53°27'50.450"	10°1'4.510"
WRBD	Ballynakill	Bundorragha (River)	53°36'26.608"	9°45'5.152"
WRBD	Ballynakill	Bunowen (River)	53°46'18.407"	9°49'3.275"
WRBD	Ballynakill	Carna (River)	TBC	TBC
WRBD	Ballynakill	Carrowbeg (River)	53°48'4.519"	9°32'9.249"
WRBD	Ballynakill	Carrownisky (River)	53°43'44.732"	9°53'27.875"
WRBD	Ballynakill	Culfin (River)	53°36'24.620"	9°53'27.775"
WRBD	Ballynakill	Dawros (River)	53°34'18.364"	9°57'42.528"
WRBD	Ballynakill	Doohulla (River)	53°24'34.008"	10°2'9.599"
WRBD	Ballynakill	Erriff (River)	53°37'0.097"	9°40'17.319"
WRBD	Ballynakill	Lettermuckno	53°18'13.595"	9°33'23.671"
WRBD	Ballynakill	Owenglin (River)	53°29'14.547"	10°1'20.707"
WRBD	Ballynakill	Owennadornaun (River)	53°40'9.240"	9°54'5.504"
WRBD	Ballynakill	Owenwee (Belclare) River	53°46'46.787"	9°34'51.128"
WRBD	Bangor	Bellagarvaun	53°58'30.305"	9°47'39.914"
WRBD	Bangor	Bunnahowna	53°54'7.777"	9°44'30.690"

Table A3.1 (continued)				
River basin district	Fishery district	Fisheries system	Latitude (N)	Longitude (W)
WRBD	Bangor	Carrowsallagh	53°53'47.120"	9°37'37.044"
WRBD	Bangor	Dooega (River)	53°55'15.313"	10°1'26.022"
WRBD	Bangor	Glenamoy (River)	54°14'39.456"	9°42'54.836"
WRBD	Bangor	Gweedaney (River)	54°17'15.564"	9°45'57.960"
WRBD	Bangor	Moyour (River)	53°49'39.187"	9°35'50.599"
WRBD	Bangor	Muingnabo (River)	54°16'15.768"	9°43'22.984"
WRBD	Bangor	Newport (River)	53°53'3.148"	9°32'21.558"
WRBD	Bangor	Owenduff (Glenamong) (River)	54°4'0.907"	9°50'4.769"
WRBD	Bangor	Owengarve (River)	53°54'2.641"	9°42'8.064"
WRBD	Bangor	Owenmore (River)	52°13'47.314"	10°10'38.082"
WRBD	Bangor	Owennabrockagh (River)	53°50'59.401"	9°33'40.826"
WRBD	Bangor	Rossow (River)	53°51'50.234"	9°33'32.102"
WRBD	Bangor	Srahmore (River)	53°55'10.874"	9°34'59.140"
WRBD	Connemara	Cashla (Costello) (River)	53°16'44.609"	9°32'20.688"
WRBD	Connemara	Gowlabeg (River)	53°23'23.072"	9°46'49.958"
WRBD	Connemara	Inverbeg (Lough and stream)	53°23'26.663"	9°38'19.356"
WRBD	Connemara	Invermore (River)	53°23'26.468"	9°39'20.338"
WRBD	Connemara	Lough Carrafinla (Stream system)	53°17'53.140"	9°33'19.471"
WRBD	Connemara	Lough Skannive (Stream system)	53°19'14.331"	9°49'9.256"
WRBD	Connemara	Owengowla (River)	53°23'34.232"	9°47'18.133"
WRBD	Connemara	Owenmore (River)	54°8'9.229"	9°48'59.060"
WRBD	Connemara	Screebe	53°23'48.682"	9°33'54.890"
WRBD	Connemara	stream (L. Nafurnace)	53°22'32.600"	9°32'45.128"
WRBD	Galway	Aille (River)	53°0'31.789"	9°23'34.126"
WRBD	Galway	Clarinbridge (Clarin) (River)	53°13'39.556"	8°52'58.850"
WRBD	Galway	Corrib (River)	53°16'11.797"	9°3'18.466"
WRBD	Galway	Crumlin (River)	53°14'4.053"	9°26'29.047"
WRBD	Galway	Kilcolgan (Dunkellin) (River)	53°12'48.630"	8°52'25.255"
WRBD	Galway	Knock (River)	53°15'0.865"	9°13'27.049"
WRBD	Galway	Owenboliska	53°14'34.964"	9°18'30.618"
WRBD	Galway	Owenriff River	53°14'30.344"	9°21'52.104"
WRBD	Sligo	Ballysadare (River)	54°12'52.173"	8°30'44.839"
WRBD	Sligo	Drumcliff (River)	54°19'41.698"	8°30'2.058"
WRBD	Sligo	Garvogue (River)	54°16'20.691"	8°28'27.287"
WRBD	Sligo	Grange (River)	54°23'34.488"	8°32'39.067"
WRBD	Sligo	Willsborough (Stream)	54°16'58.917"	8°28'36.072"
TOTAL				256

Table A3.2. Population trends in monitored river systems in Ireland with sea trout and fish counter/trapping facilities.

River basin district	Fishery district	Fisheries system	Population trend since 2005
ERBD	Drogheda	Boyne (River)	No apparent trend
ERBD	Dublin	Liffey (River)	Insufficient data
NBRBD	Dundalk	Dee (River)	No apparent trend
NBRBD	Dundalk	Fane (River)	No apparent trend
NBRBD	Dundalk	Glyde (River)	Insufficient data
NWRBD	Ballyshannon	Eany (Water)	No apparent trend
NWRBD	Ballyshannon	Eske (River)	No apparent trend
NWRBD	Letterkenny	Clady (River)	Insufficient data
SERBD	Wexford	Slaney (River)	-
SHRBD	Limerick	Feale (River)	Insufficient data
SWRBD	Cork	Bandon (River)	-
SWRBD	Kerry	Currane (Waterville) (River)	-
SWRBD	Kerry	Blackwater	Insufficient data
SWRBD	Kerry	Maine (River)	-
SWRBD	Lismore	Blackwater (River)	Insufficient data
WRBD	Ballynakill	Bunowen (River)	-
WRBD	Ballynakill	Culfin (River)	Insufficient data
WRBD	Ballynakill	Dawros (River)	Insufficient data
WRBD	Ballynakill	Erriff (River)	No apparent trend
WRBD	Bangor	Owenduff (Glenamong) (River)	No apparent trend
WRBD	Bangor	Owenglin (River)	Insufficient data
WRBD	Bangor	Owenmore (River)	-
WRBD	Bangor	Srahmore (River)	No apparent trend
WRBD	Connemara	Cashla (Costello) (River)	Insufficient data
WRBD	Connemara	Invermore*	Insufficient data
WRBD	Connemara	Owengowla* (River)	Insufficient data
WRBD	Connemara	Owenmore (River)	No apparent trend
WRBD	Galway	Corrib (River)	No apparent trend
WRBD	Galway	Owenboliska	No apparent trend
WRBD	Sligo	Ballysadare (River)	No apparent trend
WRBD	Sligo	Garvogue (River)	Insufficient data
TOTAL			31

* only a remnant population exists since a stock collapse in the late 1980s.