



European Inland Fisheries
and Aquaculture
Advisory Commission

Ecological and economic impacts of predators on freshwater fish resources and aquaculture in Europe

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Preparation of this document

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Summary

Across Europe, populations of several fish-eating birds and mammals have recovered substantially in recent decades, reflecting successful conservation measures and legal protection. This is an important biodiversity achievement. At the same time, increasing abundance and expanding distribution of some predators have intensified interactions with freshwater fish populations, inland fisheries and aquaculture. These interactions are particularly important where fish stocks are already depleted, where fish are concentrated during migration or stocking, and in aquaculture and intensively managed fisheries where fish have limited opportunities to avoid predators.

This review assesses the ecological and economic impacts of the principal fish-eating birds, mammals and piscivorous fish affecting European freshwater fish resources, fisheries and aquaculture. It also reviews available mitigation measures and identifies priorities for future policy and management. The review shows that predation should not be considered in isolation. Fish populations across Europe are being affected by habitat degradation and fragmentation, barriers to migration, altered water flows, pollution, invasive species and climate change. However, in some locations and for some fish populations, the additional mortality from predators can substantially reduce abundance, constrain recruitment and prevent recovery following habitat restoration or other management interventions.

The scale and nature of predation differ considerably among predator species. Great cormorants have the widest geographical impact because of their abundance, mobility and ability to exploit inland, coastal and aquaculture systems. Cormorant predation can be particularly significant for depleted river fish populations, shallow lakes, stocked fisheries and migrating salmonids. Mergansers can cause substantial losses of juvenile salmonids, particularly during concentrated downstream smolt migrations. Herons and egrets generally cause more localized impacts, especially at fish farms and shallow ponds. Otter impacts are also strongly site-specific, but can be severe at aquaculture facilities, stocked fisheries and small enclosed water bodies. Seals can affect coastal fisheries, salmon migrations and cage aquaculture. Introduced piscivorous fish can also adversely affect native fish communities and biodiversity where they become established outside their natural range.

Predation can affect aquatic ecosystems beyond the direct removal of fish. Selective removal of some fish species and size classes can modify fish community structure, predator–prey relationships and food webs. In vulnerable systems, high predation pressure can undermine conservation and restoration efforts for threatened fish populations. European biodiversity policy thus needs to recognize that fish is an important component of the biodiversity. Successful conservation of protected predators should be balanced with the conservation requirements of fish populations and the ecological functioning of freshwater ecosystems.

The economic consequences of fish predation in Europe are substantial. The available information indicates that predator-related losses caused by cormorants in freshwater aquaculture in the European Union exceed EUR 138 million annually, while losses to aquaculture across Europe are estimated at about EUR 250 million per year. Pond farms are particularly vulnerable, commonly reporting losses of 10–30 percent of fish stocks and substantially higher losses in severe cases. Cormorant predation is estimated to cause annual losses of approximately EUR 150–220 million to recreational and other inland fisheries through reduced fish abundance, catches and stock productivity. Significant losses, which add up to tens of millions of euros, are also reported from otters, herons, pygmy cormorants and other predators, although available information does not yet cover all European countries.

These estimates do not capture the full economic burden. Fish farmers and fishery managers also incur costs associated with injured fish, reduced growth, repeated stocking, fencing and netting, guards, scaring devices, maintenance and administrative procedures. Predation can reduce the benefits of investments in habitat restoration and fish stocking and may contribute to declining profitability or closure of vulnerable fish farms. Reduced fish abundance can also affect recreational angling, commercial fisheries, tourism and associated rural economic activities. The costs of predator conservation are therefore unevenly distributed: society benefits from wildlife conservation, while individual fish farmers, fishery managers and angling organizations often bear much of the associated economic cost.

A wide range of measures is used to reduce predator impacts. These include scaring, exclusion devices, predator-proof fencing and netting, provision of fish refuges, habitat modification, changes in stocking practices and, where permitted under national and European legislation, targeted lethal control or reproductive interventions. Such measures can reduce losses in particular circumstances, especially in enclosed aquaculture facilities. However, their effectiveness is often limited by cost, practical constraints and predator habituation. In open waters, rivers and large lakes, exclusion is generally impractical. Scaring commonly displaces predators rather than reducing overall predation, while local population control may have only temporary effects where mobile predators rapidly recolonize managed areas.

The large number of derogations already issued under the EU Birds and Habitats Directives demonstrates both the extent of predator conflicts and the limitations of predominantly local responses. More than 4 000 derogations relating to piscivorous birds were issued between 2020 and 2025, alongside numerous derogations for protected mammals. Their implementation varies considerably among countries and is generally not coordinated across the geographical ranges of mobile predator populations. Local actions alone are therefore unlikely to resolve conflicts involving highly mobile species such as the great cormorant.

The review reaches the following main conclusions:

- predation is an important ecological and economic pressure on freshwater fish, fisheries and aquaculture in parts of Europe and should be considered together with other pressures affecting fish populations;
- impacts are highly context-specific but can be particularly severe for depleted or threatened fish populations, migrating fish, stocked fisheries and open aquaculture systems;
- the economic costs extend well beyond the value of fish consumed and include prevention, restocking, infrastructure, lost production, reduced fishing opportunities and wider rural economic impacts;
- existing mitigation measures are useful, but have generally not provided a durable solution to widespread conflicts involving mobile predators;
- existing management is fragmented among countries and between nature conservation, fisheries, aquaculture and water-management authorities; and
- conservation policy needs to balance the favourable recovery of predator populations with the protection of freshwater fish biodiversity and the social and economic sustainability of fisheries and aquaculture.

A more coordinated European response is therefore required along the following actions.

- 1) Predation should be explicitly incorporated, where substantial, into assessments and recovery programmes for freshwater fish populations alongside habitat, water-quality, connectivity, climate and other pressures.
- 2) Implementation of the Birds and Habitats Directives should provide clear, proportionate and timely mechanisms for addressing demonstrated serious ecological or economic damage, while maintaining the conservation objectives of those Directives.
- 3) Management should operate at the geographical scale of the predator population. Highly mobile species require coordinated regional or European approaches. In particular, the EIFAAC Framework for a European Management Plan for the Great Cormorant provides a basis for coordinated action. More localized predator conflicts should be addressed through national or regional action plans with clearly defined responsibilities, intervention criteria and management options.
- 4) Governments should consider transparent mechanisms for compensating verified serious losses and supporting effective preventive measures where the costs of protected wildlife are disproportionately borne by individual aquaculture enterprises or fishery organizations. Compensation should complement, rather than substitute for, effective management.
- 5) Stronger institutional coordination is required between authorities responsible for nature conservation, fisheries, aquaculture and water management. Fisheries and aquaculture institutions should participate fully in decisions concerning the management of fish-eating predators, together with research organizations and relevant stakeholder groups.
- 6) Monitoring and research should be focused on supporting management decisions. Harmonized information is needed on predator abundance, fish population status, economic losses, and the effectiveness and costs of measures and management interventions. Research should address crucial evidence gaps, such as economic damages, but lack of complete scientific certainty should not prevent proportionate management action where serious ecological or economic impacts have already been adequately demonstrated.

A balanced European approach should seek neither unrestricted predator control nor protection of predator populations without regard to their wider consequences. The objective should be the conservation and sustainable management of the entire freshwater ecosystem and its associated natural capital, including fish biodiversity, protected predators, viable fisheries and aquaculture, and the social and economic benefits these resources provide.

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1 Introduction

Across Europe, fish-eating predators—particularly birds and mammals—have increased in abundance due to conservation successes and regulatory protection (European Environment Agency, 2026). While positive from a biodiversity perspective, these developments have also led to considerable interaction and conflict with freshwater fish resources and aquaculture operations. Collectively, diversity gains driven by conservation objectives and trajectories for one or a few species do not necessarily translate into ecological stability, highlighting an urgent need to explicitly incorporate species conservation objectives into frameworks evaluating diversity and economic stability of all species – both predators and prey. This need is driven by the alarming number of threatened freshwater fish species in Europe: 42% of European freshwater fish are threatened with extinction, and an additional 18% classified as Near Threatened (Ford et al., 2026). In addition, 85% of freshwater fish listed under the Habitats Directive are classified as Unfavourable-Inadequate or Unfavourable-Bad¹.

In the light of these challenges and growing conflicts between recovering conservation species and fish biodiversity conservation, and fisheries and aquaculture enterprises, there is critical need to update and improve understanding of the impacts of fish predators on inland fish and aquaculture resources, and tease out impacts of other anthropogenic pressures to balance conservation objectives with sustainable fisheries and aquaculture management. Balancing conservation objectives with sustainable fisheries and aquaculture management requires improved evidence, dialogue and coordinated action. This review synthesises information on fish predators in inland and coastal waters and aquaculture across Europe, focusing on ecological and economic dimensions, and implications for sustainable fisheries and aquaculture, and to develop evidence-based recommendations to support effective management and policy responses.

1.1 Objectives

The objectives of this working paper are to:

- Review ecology and ecological and economic impacts of fish predators on freshwater and coastal fish resources, fisheries and aquaculture in Europe.
- Summarise mitigation measures to address the impacts of fish predators on freshwater and coastal fish resources, fisheries and aquaculture in Europe.
- Outline potential actions to overcome ecological and economic impacts of predation on freshwater and coastal fish, fisheries and aquaculture in Europe.

The intention here is to review only predators that are the subject of human-wildlife conflicts either because of considerable ecological or economic impact on fish or fisheries and aquaculture or preclude recovery of aquatic biodiversity following remedial environmental interventions. It is acknowledged that fisheries are exploited by society for food, income generation and leisure, which in some cases may lead to declines in stocks, but this scenario is primarily related to marine fisheries and rarely intersects with inland waters where multiple fish-wildlife conflicts occur. Indeed where fisheries exploitation is found in inland waters, fishing intensity tends to be well-regulated or the fish are returned to the water live after capture (i.e. recreational fishing) (Arlinghaus et al., 2021).

¹ nature-eionet.europa.eu/article17/species/progress/?period=6&group=Fish&conclusion=overall+assessment

The report is founded on the extensive review carried out for the EIFAAC Framework for a European Management Plan for the Great Cormorant (FAO, 2026a) and is expanded to other fish predator species through a review of the scientific literature using Web of Science, SCOPUS and Google Scholar. Information on the status, distribution, abundance and ecology of bird species was collated from the European Environment Agency European Nature Information System (EUNIS)², BirdLife International and Wetlands International Critical Site Network Tool 2.0³, European Breeding Bird Atlas⁴ and the BirdLife Datazone⁵. Data on the number of derogations under Article 9 of the Birds Directive and Article 12 of the Habitats Directive were compiled from “Overview of derogations and exceptions to species protection across the EU for the period 2020-2025 (EEA)”⁶. It should be noted, information on economic impacts is limited and largely restricted to the primary study carried out by EIFAAC/FEAP/EAA (FAO, 2026b). Details of the distribution and abundance, ecology, ecological and economic impact of predation and management measures are compiled for each species in a series of annexes and summarised in the main report. Note, it is not the intention to repeat the comprehensive review of the impact of cormorants in inland fisheries and aquaculture in Europe. Readers are referred the EIFAAC Framework for a European Management Plan for the Great Cormorant (FAO, 2026a) as a complementary text to this report.

2 Status of European inland fisheries and aquaculture

Freshwater ecosystems in Europe are highly diverse but under substantial pressure from habitat loss and fragmentation, river regulation and dams, pollution and eutrophication, overexploitation, invasive non-native species and climate change. These pressures are directly relevant to the objectives of the EU Water Framework Directive (2000/60/EC), which establishes a framework for protecting and restoring the ecological status of Europe’s surface waters. For rivers and lakes, fish communities are one of the biological quality elements used to assess ecological status, meaning that changes in fish abundance, composition and age structure can provide evidence of ecological status.

The most recent State of European Waters (European Commission, 2025; European Environment Agency, 2024) WFD assessment demonstrates the continuing degradation of European freshwater ecosystems. Data from the third River Basin Management Plans⁷ indicate that only approximately 38–40% of European surface waters achieve good or high ecological status, with little overall improvement since 2010. The principal pressures include hydromorphological alteration and river fragmentation, pollution and nutrient enrichment, water abstraction, altered hydrological regimes, invasive species and climate change. These pressures directly affect fish populations by reducing habitat quality, disrupting migration and spawning, altering food availability and increasing mortality and the poor condition of many freshwater fish populations is a major contributor to failing good ecological status or potential.

The latest assessment under the EU Habitats Directive (92/43/EEC) provides a similar picture and shows the overall situation remains poor with approximately 63% of species protected under the

² <https://eunis.eea.europa.eu/species>

³ <https://criticalsites.wetlands.org/en/species>

⁴ <https://ebba2.info/maps/>

⁵ <https://datazone.birdlife.org/search>

⁶ <https://www.eea.europa.eu/en/analysis/maps-and-charts/overview-of-derogations-and-exceptions-dashboards>

⁷ https://environment.ec.europa.eu/topics/water/water-framework-directive_en#state-of-play-of-3rd-rbmp-adoption-in-eu-27

Habitats Directive having unfavourable conservation status. This is of particular relevance because freshwater fish are amongst the most threatened species group both globally (Sayer et al., 2025) and across Europe (Ford et al., 2026). Migratory species such as Atlantic salmon, European eel and several potamodromous fishes are particularly vulnerable because barriers and changes in river connectivity disrupt access to spawning and feeding habitats.

The Convention for the Protection of the Marine Environment of the North-East Atlantic (the “OSPAR Convention”) also recognises that cod, eel and Atlantic salmon are under threat or declining in OSPAR marine areas, which include all European coastal fisheries (McLeod, 2023). This supports NSAC and BSAC (NSAC/BSAC, 2024) and Jepsen et al. (2026) findings that cod and other coastal species are under considerable pressure from cormorant and seal predation in the North Sea and Baltic Sea regions.

The poor overall status of European surface waters represents a fundamental constraint on the long-term sustainability of many freshwater and diadromous fish species and associated inland fisheries. To improve this status, there is continuous dialogue to integrate fisheries management into discussions on habitat restoration, river connectivity, environmental flows, water-quality improvement and conservation of fish biodiversity. Yet one pressure that is largely overlooked is the role of predation.

Whilst inland waters in Europe are potentially constrained by pressures on aquatic ecosystems, catches are relatively stable in individual countries across the continent (Arthur, 2025; Cowx, 2015). Total catch has fallen slightly from 165 346 tonnes in 2020 to 154 591 tonnes in 2024, with catch dominated by Turkey, Ukraine, Germany, Poland and Finland (FAO, 2026c). The estimated value of inland fisheries in Europe in 2024 was approximately EUR 74 million, which is marginal compared with the value of recreational fishing and aquaculture (Arthur, 2025; FAO, 2026b; Nyboer et al., 2022). It should be noted this does not include the value of coastal fisheries that are also impacted by predation by piscivorous birds and mammals (e.g. Harris et al., 2008; Jepsen et al., 2018, 2026; Lacombe et al., 2025; NSAC/BSAC, 2024; Ovegård et al., 2021).

Freshwater aquaculture in Europe is dominated by a few key species – trout, carps and Atlantic salmon – grown outdoors both intensively in raceways, tanks and nets and extensively in ponds, plus intensively indoors in recirculating systems. Atlantic salmon is also grown intensively in coastal cages in inshore waters. It is the outdoor systems that are vulnerable to predation by piscivorous birds, especially cormorants, herons and egrets, and otters and mink, and, in the case of coastal salmon cage production, by seals.

Whilst salmon farming has increased in coastal waters over the past two decades, freshwater aquaculture in the EU has been relatively stable around 300 000 tonnes, valued at between EUR 1.11 and 1.45 billion annually between 2020 and 2024 (EUMOFA, 2025⁸). The same marginal increase in production is also evident when non-EU countries are included, producing between 480 000 and 490 000 tonnes annually between 2020 and 2024, although the value has increased from EUR 1.42 to 2.08 billion annually over this period (FAO 2026b). This static situation is despite global inland aquaculture production increasing dramatically over the last decade (FAO 2026c), and considerable investment in aquaculture development by the European Commission. Part of the problem for the static nature of aquaculture is the high input costs for production, but also predation by protected birds and otters has increased considerably and reached a level at which many pond farms in several regions are faced with heavy economic losses and are no longer operational (FAO, 2026b).

⁸ <https://fishery-aquaculture-market-observatory.ec.europa.eu/en/data/dashboards/aquaculture>

One of the reasons for predators not being fully recognised as a “problem” is because they are not categorised as a pressure under the Water Framework Directive⁹ or Habitats Directive¹⁰, therefore predation is not required to be monitored by Member States. Predation is thus largely overlooked by major pressures such as habitat modification or pollution. The significance of predators impacting inland fish and fisheries in Europe is, however, well-illustrated in national assessments of the impact of multiple stressors on Atlantic salmon populations in Europe by the ICES Working Group on Effectiveness of Recovery Actions for Atlantic salmon using the classification system developed by the Norwegian Atlantic Salmon Committee (Forseth et al., 2017). The system categorises the major threats to salmon populations, and evaluates if the threats are stabilized or developing. Twelve threats or stressors are evaluated (Pollution, Barriers; Water Regulation, Exploitation, Aquaculture, Habitat Degradation, Diseases/ Parasites, Climate Change, Invasive species, Stocking, Predators and Other). The methodology ranks the overall magnitude of the stressor as it currently impacts fish stocks and population viability, taking into account the severity of the stressor, how widespread it is nationally and how much has been done to effectively mitigate its impacts on one dimension – the effects axis. The second dimension, the development axis, describes the likelihood for further reductions in population size or loss of additional populations in the future, taking into account the likelihood of worsening or the likelihood of effective mitigation measures. Combined, these axes forms a 2D continuous classification system in which the impact factors can be categorized into four major trajectories (Figure 1), the most prominent of which is the upper right where factors are affecting populations to the extent that populations may be critically endangered or lost in nature and that have a high likelihood of causing even further reductions. Here, current mitigation measures are unable to hinder expansion of negative impacts in the future.

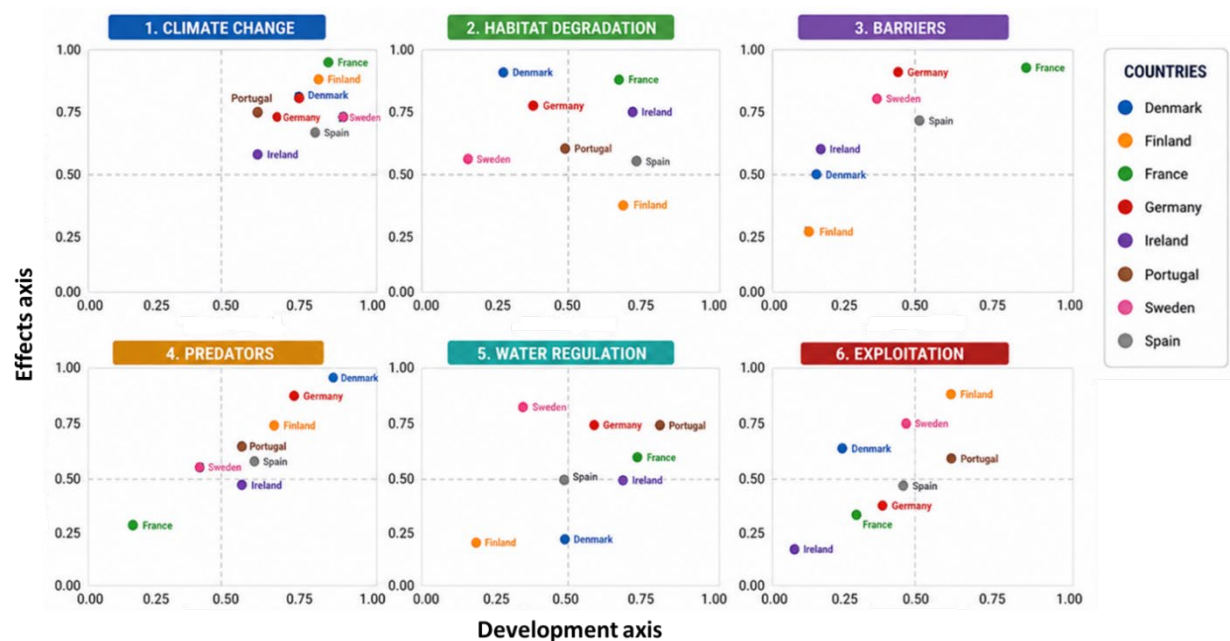


Figure 1. Assessment of the impact of multiple stressors on Atlantic salmon populations in Europe

The most prominent and relevant stressors that impact each EU jurisdiction are Barriers; Water Regulation, Exploitation, Habitat Degradation, Climate Change and **Predators**, and these were plotted for European countries supporting Atlantic salmon (Figure 1). Alongside climate change, predators are

⁹ <https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A02000L0060-20260510&qid=1786886683299>

¹⁰ <https://eur-lex.europa.eu/eli/dir/1992/43/oj/eng>

the most prevalent stressors both now and into the future – top right boxes. Cormorants, goosanders, egrets, otters, pikeperch and catfish are the most problematic predators in inland waters, while seals and porpoises are prominent in coastal areas, although their significance varies between jurisdictions. Nevertheless cormorants rank as the most predominant predator in many countries.

There is no doubt that predation is another fundamentally important stressor of inland fish and fisheries, and aquaculture systems, especially predation by piscivorous animals such as the great cormorant (*Phalacrocorax carbo carbo* and the subspecies *Phalacrocorax carbo sinensis*), goosander (*Mergus merganser*) and Eurasian otter (*Lutra lutra*). The problem is now becoming, yet again, a subject of considerable debate in political circles notably Member of the European Parliament asking questions of the European Commission¹¹ to address the cormorant fish conflict and Baltic Sea Advisory Council (BSAC¹²) and North Sea Advisory Council (NSAC) (BSAC, 2025; NSAC/BSAC, 2024) preparing a statements on the management of cormorants and seals to mitigate the impact of their predation on fisheries and aquaculture.

Against this backdrop, however, there are considerable scepticism that predation is a substantial pressure and additional mortality from predators is only locally important. The point raised is that existence of substantial predation does not by itself demonstrate that predators are responsible for poor fish status or declining fisheries where fish populations are already stressed by habitat degradation, barriers, pollution, climate change, fisheries exploitation or other pressures.

To evaluate this impasse on the role of piscivorous animals on fish biodiversity, fisheries and by extension aquaculture, the following sections review the ecological and economic evidence underpinning the impact of predation on these resources, inherent biodiversity and livelihoods in Europe.

3 Ecological impacts of predation on inland fish, fisheries and aquaculture

3.1 Piscivory and trophic dynamics

Natural mortality is a fundamental ecological process structuring fish species composition, population and community dynamics, and ecosystem functioning (Begon et al. 2006; Hilborn and Walters, 1992; Wootton, 1999). Natural mortality occurs in many forms, including starvation, parasites and disease, climatic stressors but one of the main sources is predation (Figure 2). Whilst it is critical to tease out the role of these other sources on natural mortality, essentially through life tables and similar tools, this report focusses explicitly on the role of predation.

¹¹ [https://www.europarl.europa.eu/RegData/etudes/BRIE/2026/782619/EPRS_BRI\(2026\)782619_EN.pdf](https://www.europarl.europa.eu/RegData/etudes/BRIE/2026/782619/EPRS_BRI(2026)782619_EN.pdf)

¹² <https://www.bsac.dk/wp-content/uploads/2024/12/BSACrecommendation-on-cormorants.pdf>

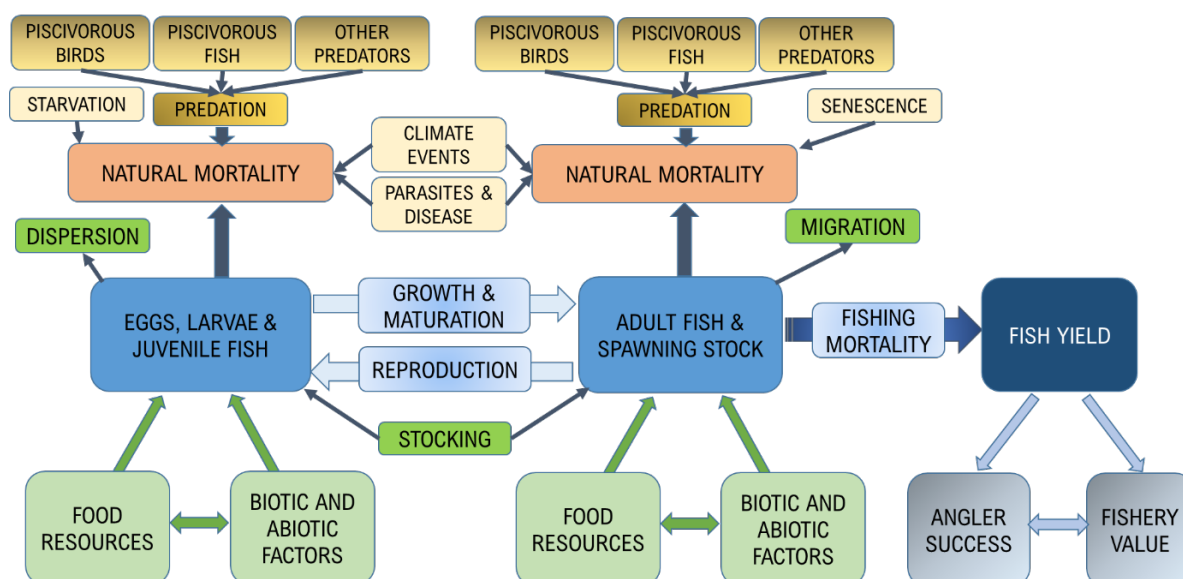


Figure 2. Chart to illustrate the complexity of the factors that influence fish populations in rivers (after Cowx, 2003)

Predation is an integral ecological process that regulates species abundance, diversity and community structure through trophic interactions (Begon et al.; 2006 Wootton, 1999). Factors such as food availability, breeding habitat and predation are considered limiting when they prevent population growth or cause population declines (Newton, 1998). The influence of predation on prey populations depends largely on how predators respond to changes in prey density (Begon et al., 2006). Under natural conditions, predator-prey populations typically fluctuate in dynamic equilibrium. However, when predator abundance becomes disproportionate to prey availability, predation pressure can exceed sustainable levels, leading to asynchronous predator-prey dynamics, declines in vulnerable prey species, and broader destabilization of ecosystem structure and function (Holling, 1959).

In Europe, inland fish are subject to predation from a diverse assemblage of taxa, including fish, mammals, such as the European otter (*Lutra lutra*) and seals (e.g. *Phoca vitulina*, *Halichoerus grypus*), and birds, such as the grey heron (*Ardea cinerea*), mergansers (*Mergus* spp.), and the great cormorant (*P. carbo carbo* and *P. carbo sinensis*). While these predators differ substantially in ecology and behaviour, they are often grouped together in debates surrounding fisheries and aquaculture impacts. Understanding these predator-prey dynamics is essential for fisheries management and conservation policy, and more so because of the growing concern about impact of predation on fish stocks and fisheries, and aquaculture production systems. This arises because fish-eating predators—particularly birds and mammals—have increased in abundance and expanded their distribution ranges across Europe due to conservation successes and regulatory protection. While positive from a biodiversity perspective, these developments have elevated ecological and economic pressures on freshwater and coastal fisheries resources and biodiversity, and aquaculture enterprises.

Whilst predation is a natural ecological process (Figure 2), conflicts arise when numbers of predators exceed those considered desirable to maintain the abundance and distribution of prey species, especially species of conservation concern, or where predators exploit commercial enterprises, such as put-and-take fisheries and aquaculture facilities. Problems also arise where predators reduce the capacity of vulnerable prey populations to recover naturally, either because of continuous intense predation impacting recruitment processes or predation curtailing recovery of severely depleted populations. These scenarios are prevalent in inland and coastal fisheries and aquaculture settings and are becoming more widespread following the recovery of several bird and mammal species, especially

the great cormorant, heron, mergansers and otter, over the past 20-30 years after receiving protected status under various European and national initiatives such as the Bern Convention, Birds Directive and Habitats Directive. These single species conservation successes raise the potential for conflict, especially where commercial ventures are impacted or biodiversity is compromised.

Against this evolving picture, others argue that there is little or no evidence for impact on fisheries and the root causes are overfishing or habitat degradation (e.g. Birdlife International¹³ and Wetland International¹⁴). There is, however, little direct evidence of overfishing in inland waters (Allan et al., 2005) and despite considerable investment to improve aquatic habitats under the HD and WFD, many fish populations have not recovered or are constrained from recovery by predation (Falkegård et al., 2023; Smout et al., 2014). Furthermore, avian predation, in particular, has resulted in degradation of ecosystems and disruption of ecological functioning through damage to trees and the underlying ground from nesting and roosting birds (Alves and Moura, 2021; Goc et al., 2005) and disruption of native bird species (Verlin, et al., 2026).

3.2 Ecological impacts of piscivorous birds

Birds represent a highly visible and ecologically significant group of fish predators. EIFAC (1988) listed 29 species of interest, but many are only temporary visitors to Europe, found in small numbers or are scavengers or predominantly predate on marine fishes. This summary focuses on species that are known to directly predate on freshwater fish open water bodies and fish farms (Annexes 8.1-8.10) and have come under intense scrutiny in Europe, especially after their successful recovery following improved conservation protection under the Bern Convention and EU Birds Directive. Details of the distribution, ecology, including feeding behaviour, ecological and economic impacts along with impact mitigation measures are provided in Annexes 8.1-8.10, and the ecological components are summarised in Table 1.

There is a well-established and growing body of evidence to illustrate the ecological impact of fish-eating birds on fisheries and aquaculture – see reviews under EU Cost projects REDCAFE and INTERCAFE. The Horizon Europe project ProtectFish¹⁵, the EIFAAC review under the Framework for a European Management Plan for the Great Cormorant (FAO 2026a), literature such as Kindermann (2008), and Cowx (2013), and information provided in Annexes 8.1-8.10. The direct ecological impact on fish populations depends on different feeding strategies, different types of water bodies exploited, different species and size of prey available and most notably the number of birds, their distribution and migratory capabilities and behaviour.

Feeding strategies vary considerably among predator groups. The grey heron and egrets are a solitary, shallow-water ambush predator, mergansers are pursuit-diving specialists in clear river systems, and cormorants are highly adaptable, often gregarious predators capable of exploiting a wide range of habitats. Cormorants and herons also consume larger prey than mergansers and grebes, the latter being constrained by beak size. As a consequence, grey herons tend to exert fine-scale, habitat-constrained pressures, mergansers tend to show seasonal, river-focused predation, while cormorants produce spatially concentrated, but potentially large-scale impacts, particularly in when feeding in aggregations.

¹³ <https://www.birdlife.org/news/2025/09/23/mass-culling-of-cormorants-wont-save-fish/>

¹⁴ <https://europe.wetlands.org/our-call-to-resolve-the-cormorant-fisheries-conflict-based-on-proper-science-respecting-eu-and-international-laws-and-processes/>

¹⁵ <https://protectfish.eu/>

Table 1. Summary of principal bird species feeding on fish in European inland waters

Species	Distribution	Number of mature individuals ‡	Habitats	Movements	Diet	IUCN Red List category and population trend
Great cormorant <i>Phalacrocorax carbo carbo</i> and <i>Phalacrocorax carbo sinensis</i>	Coastal areas of N and W Europe. Also central and SE Europe	803 000-1 020 000 (2013) but estimated range 1.5-2 million in 2025	West and northern Europe, largely coastal and inland waters. East Europe breed near large inland waters	Highly migratory/partially migratory/dispersive, according to population but movements between northern breeding areas and southern wintering habitat	Almost exclusively fish. In inland waters feed on eel/roach/ruffe/pike-perch/bream /perch/rudd/tench/salmonids	Least concern
Pygmy cormorant <i>Microcarbo pygmaeus</i>	Restricted range. Occurs small no. in Croatia/Greece/Romania/Turkey	53 300-134 000	Open standing or slow flowing freshwater/ swamps /floodplains and coastal wetlands	Migratory/partially migratory/resident populations	Chiefly fish including rudd/pike/carp/roach/loach /bitterling/tench	Least concern
Grey heron <i>Ardea cinerea</i>	Widely distributed mid-latitudes and coastal areas of Scandinavia	447 000-782 000	Mainly lowland (<500 m) Prefers lotic/lentic shallow fresh waters, with trees close by	Migratory/partially migratory/dispersive. Most European populations overwinter in southern/western areas	Variable. - mainly fish (1–25 g), amphibians, small mammals	Least concern
Little egret <i>Egretta garzetta</i>	Very patchy distribution SW and E Europe, the Black Sea and Eastern Mediterranean	121 000-190 000	Usually in lowland shallow lakes/pools/gently flowing rivers. Sometimes coastal areas	Migratory/dispersive. Majority overwinter in North Africa	Mainly small (12–15 cm) amphibians, insects, fish (tench/carp/roach/loach)	Least concern
Great white egret <i>Ardea alba</i>	Western central and southern Europe. Rare, breeding populations in USSR/Turkey/Hungary/Czechia	No specific European estimate. Global population 1 600 000-2 400 000	Largely restricted to extensive wetlands and margins of fresh water in lowland regions	Partially migratory/dispersive. Overwinters around Adriatic/Mediterranean	In wet season, chiefly fish (sunfish/carp/tench/trout). In dry season insects	Least concern

Goosander <i>Mergus merganser</i>	Widely distributed in N. Europe, especially UK and Scandinavia	134 000-206 000	Upper basins of rivers/lakes in forest/mountain areas in close proximity to mature trees	Migratory/partially migratory. Overwinters in NW Europe/Black Sea	Chiefly fish, with some preference for salmonids	Least concern
Red-breasted merganser <i>Mergus serrator</i>	Widely distributed in N. and western Europe, especially UK and Scandinavia	140 000-240 000	Wide range, including oceanic/temperate forest/estuarine. Wintering birds prefer marine habitats	Migratory/partially migratory. Most overwinter in S. Europe	Primarily fish (salmonids/eels/lampreys/minnows/carp)	Least concern
Great-crested grebe <i>Podiceps cristatus</i>	Widely distributed in most of Europe, except for north	609 999 – 997 000	Cool-cold standing freshwaters, natural/artificial + fishponds	Migratory/dispersive. Shifts from inland waters to the sea in winter	Chiefly fish, including roach/bleak/gudgeon/perch/salmonids	Least concern
Dalmation pelican <i>Pelecanus crispus</i>	SE Europe, especially Romania/Bulgaria/Albania/Turkey/Croatia/Caucasus	11 400-13 400	Similar to often shared with <i>P. onocrotalus</i> also tolerates hilly terrain with small open waters	Migratory/partially migratory	Entirely fish, including carp/perch/asp/roach/pike/tench	Least concern
White pelican <i>Pelecanus onocrotalus</i>	SE Europe/Romania/Greece/Turkey/ Caucasus	4 900-5 600	Low-lying/shallow warm water bodies/river deltas/wetlands	Migratory/non-dispersive movements between Mediterranean and East Africa	Almost exclusively fish + roach/carp/bream/rudd/bleak/pike	Least concern

‡Source: BirdLife International (2015, 2021); <https://datazone.birdlife.org/search>

Ecological pressures are also modified by the breeding and migratory behaviour of the bird species. Cormorants are highly migratory and exert predation pressures on water bodies across their migratory range at different times of the year, thus they have widespread foraging impacts. Herons/egrets and mergansers tend to be more resident so predation is more localised. These differences can have considerable implications for managing populations and regulating depredation.

Predation by cormorants in Europe, particularly the great cormorant (*P. carbo carbo* and *P. carbo sinensis*), has attracted substantial scientific and political attention (Kindermann 2008; Cowx 2013; FAO 2026a, b). Following population recoveries in the late twentieth century, cormorants have expanded their range and abundance across Europe, increasingly exploiting inland waters. Their highly efficient feeding and tendency to form large foraging aggregations have made them central to debates concerning fish population declines and fisheries management. Nevertheless, even rare visits by cormorants can have serious consequences for wild river fish populations such as salmon, marble trout, brown trout, grayling, barbel and nase (FAO 2026a and references therein). Trout and salmon smolts are particularly vulnerable to cormorant predation, as are stocked, hatchery-reared trout and other commonly stocked species. Cyprinid fishes (e.g. roach, bleak and bream), northern pike and perch-like fish (pikeperch and perch) are also at great risk of being adversely affected by cormorants (FAO 2026a and references therein). Depletion of fish stocks in lakes as a result of predation by cormorants has been documented in many countries across Europe and the impacts are particularly severe in small shallow lakes (FAO 2026a and references therein).

A similar situation is seen in coastal waters, where cormorants consume a similar or greater proportion of fish than is caught by commercial fishing (Hansson et al., 2017). Jepsen et al. (2026), using tagging studies, showed cormorants have the potential to consume between 52% and 88% of cod, depending on cod size, and between 14% and 84% for the least and most vulnerable size classes of flounder, respectively.

Cormorants are particularly efficient and flexible predators and considering a total population of 1-5 to 2 million birds in Europe (as a geographical area, rather than solely the European Union) and the fact that they each consume approximately 500 g of fish/day (Grémillet et al., 1996, Keller and Visser, 1999), equates to ≈365 000 tonnes of fish consumed each year, assuming they all forage within European waters the whole year.

These studies highlight the considerable ecological impact great cormorants can have on fish and fisheries in inland and coastal waters. They also predate heavily on aquaculture ponds and stocked recreational fisheries where the impact is measured in an economic rather ecological context, and can be substantial (see Section 4.1).

The grey heron is one of the most widespread fish-eating birds. As a generalist wading bird, the grey heron exploits a wide range of aquatic habitats, including rivers, lakes, wetlands and fish farms. The grey heron is a solitary, visually oriented predator that typically forages in shallow water using a sit-and-wait or slow-stalking strategy. Fish typically dominate the diet of grey herons, often comprising over 80% of prey items in inland systems (Marquiss and Leitch, 1990). Common prey species include small cyprinids, perch, and juvenile salmonids. Herons generally select prey based on size and accessibility, favouring small to medium-sized fish that can be captured and swallowed whole. Grey herons exhibit flexible foraging behaviour, often exploiting habitats with high prey density. They are known to concentrate in areas such as fish farms, stocked lakes, and shallow river margins, where prey is abundant and easily accessible (Carss, 1993). In small or enclosed systems, such as fish ponds, herons can have significant local impacts, particularly on stocked fish, but in large rivers and lakes with

complex habitats and diverse fish communities, heron predation is generally considered to be marginal (Marquiss and Leitch, 1990).

The little egret and great white egret occupy similar habitat and adopt a similar feeding strategy as the grey heron. Their distribution, however, is more constrained but they can form large colonies during the breeding season. Great white egrets are effective predators of small freshwater fish and can exert substantial local predation pressure, particularly in shallow wetlands, fishponds and aquaculture systems with limited fish refugia. The greatest threats are likely to be on fish farm ponds rather than larger open water bodies.

Among avian piscivores in Europe, mergansers and grebes are of particular importance. Species such as the goosander, red-breasted merganser and great crested grebe are highly specialised fish-eating ducks that exploit rivers, lakes and estuaries. Their increasing populations in parts of Europe, combined with their reliance on fish prey, have led to growing interest in their ecological role and potential impacts on fisheries. They generally select small- to medium-sized fish, typically approximately 5–20 cm in length, particularly salmonids. Downstream migration of smolts can temporarily create concentrated and highly vulnerable prey resources. Such predation may be particularly critical where salmon populations are already depleted or exposed to multiple environmental pressures. They can cause considerable damage and potentially lead to collapse in salmon stocks when targeting smolts on their outward migration (Feltham 1995; Feltham et al. 1999; Wilson et al. 2003), or can create local conflicts with recreational fisheries when targeting other species. All three species is fully migratory in much of their range, although populations in temperate regions may undertake only relatively short-distance movements to their breeding waters throughout the year. This dissipates the predation pressure somewhat between seasons.

Pelicans have fairly restricted distribution but can breed in large colonies often foraging communally. The species is entirely piscivorous, and can consume ≈ 1.2 kg of fish per bird per day so ecological impact can be high. Pelican population dynamics are linked with the availability of local fish resources, suggesting pelicans are directly selecting abundant prey and can regulate fish availability (Pyrovetsi and Economidis, 1998). This can have impact on fish populations and lead to conflict with local fisheries.

An often-overlooked aspect of predation is the transformation of ecosystems. Predation on higher trophic level fish species such as pike and perch and alter the predation pressure on lower zooplankton-eating fish, leaving small pelagic species to proliferate (Olin et al., 2022), and these deplete the larger zooplankton that regulate algal growth (Gerke et al., 2018). Ultimately this can accelerate eutrophication processes (Donadi et al., 2017; Eklöf et al., 2020), with algal blooms causing oxygen depletion, and negatively impacting water quality and aquatic biodiversity (Alves and Moura, 2021). Dense breeding populations of cormorants can also transform forest ecosystems (Goc et al., 2005). The accumulation of guano (faeces) in these areas can lead to canopy loss of up to 90 percent in riparian forests, triggering cascading effects on other organisms, including amphibians, and there is now emerging evidence that the proliferation of great cormorants caused the loss of other bird species (Verlin, et al., 2026). Additionally, the nutrient enrichment of adjacent waterbodies from guano runoff can disrupt ecological processes, resulting in reduced biodiversity and biomass of aquatic invertebrates and plants.

3.3 Ecological impacts of piscivorous mammals

Like birds, the recovery of mammal populations has brought several species into conflict with fisheries and aquaculture, with conservation policies complicating management (e.g. otters and seals are

protected under EU directives). The distribution and abundance of otters has increased markedly over the past two decades and they are now found across Europe (see Annex 8.11), while seals have increased in abundance in coastal regions of the North Sea and Baltic Sea (see Annex 8.13) (Table 2). In addition, the non-native American mink has a wide distribution range across Europe having escaped or been released from fur farms across the continent.

Table 2. Summary of principal mammal species feeding on fish in European inland waters

Species	Distribution	Number of individuals	Population trend	Habitats Directive status IUCN Red List category
Otter	Widespread across western, central and northern Europe and Iberian Peninsula	Unknown but increasing	Rapid recovery in numbers and distribution since the 1990s	Near Threatened
American mink	Widespread across northern Europe and Baltic States and populations in France, Spain and UK	Unknown	Range increasing despite being listed as species of concern under EU invasive alien species regulation	Invasive non--native
Grey seal	North-East Atlantic Ocean	‡ > 250,000	Increasing	Favourable condition status Least Concern
Harbour seal	North-East Atlantic Ocean	‡ 120,000	Increasing	Unfavourable condition status Least Concern
Baltic ringed seal	Baltic Sea in Bothnia Bay, Gulf of Riga and Gulf of Finland	‡ 10,000	Stable	Unfavourable condition status Least Concern
Saimaa ringed seal	Lake Saimaa, Finland	‡ less than 300	Stable	Unfavourable condition status Endangered

‡ Source: NAMMCO [<https://nammco.no/seals-and-walrus/>] and Banga et al. (2022)

The European otter is a semi-aquatic carnivore that feeds predominantly on fish, although its diet is opportunistic and varies with prey availability. Otters are generally considered meso-predators within freshwater ecosystems, exerting top-down control on fish populations while also linking aquatic and terrestrial food webs (Clavero et al., 2003). Their foraging behaviour is influenced by factors such as water depth, flow conditions, and habitat structure, which affect prey accessibility.

While otters are efficient predators, most studies suggest that their effects are density-dependent and context-specific, rather than uniformly detrimental. In natural river systems with diverse fish communities, otters tend to target abundant or vulnerable individuals, potentially contributing to population regulation rather than decline (Clavero et al., 2003). However, in certain contexts—particularly in small, enclosed systems such as fish ponds or heavily managed fisheries—otter predation can have more pronounced effects. High local densities of otters, combined with limited prey refuges, may lead to significant reductions in fish stocks, particularly for commercially or recreationally valuable species, such as carps or salmonids (Kloskowski, 2005), and especially in open, intensively stocked fisheries. These situations often give rise to conflicts between conservation

objectives and economic interests. The situation is exacerbated because otters tend to “overkill” their prey often leaving dead and damaged fish beyond those that they consume. There are arguments that the otter will return later to consume these dead fish but the damage is already done.

The American mink (*Neogale vison*) is an invasive semi-aquatic predator capable of exerting substantial predation pressure on freshwater fish in Europe, particularly juvenile salmonids inhabiting small streams (Table 2). Evidence demonstrates that hatchery-reared brown trout can be more vulnerable to mink than wild fish (Vehanen et al., 2022), and that stocking may indirectly increase predation on wild conspecifics, whereas increased habitat complexity can reduce mink predation. Nevertheless, mink are generalist predators with considerable dietary flexibility, and fish may constitute only a minor component of their diet where alternative prey is abundant. Consequently, the presence or abundance of mink should not be interpreted automatically as evidence of population-level effects on freshwater fish. The strongest impacts are likely to occur where valuable or conservation-sensitive fish are concentrated, where alternative prey are scarce, and where habitat provides limited refuge. Economic impacts on aquaculture and fisheries are plausible and can be locally substantial, but contemporary Europe-wide assessments remain poorly developed (Harrison and Symes, 1989). The addition of *N. vison* to the EU list of invasive alien species of Union concern in 2025 and the demonstration that landscape-scale eradication is feasible in eastern England (Martin and Baker, 2026) provide a strong basis for targeted management. For fisheries and aquaculture sustainability, priority should therefore be given to evidence-based assessment of predator-induced mortality, habitat restoration, predator exclusion at high-value facilities, and strategically coordinated mink control where demonstrable ecological or economic impacts occur.

Seals, including *Phoca vitulina* and *Halichoerus grypus*, act as external apex predators, extending their influence into freshwater systems through estuarine and riverine foraging (Table 2). They tend to select energetically profitable prey, including larger fish such as salmonids and eels. Their predation is typically episodic and concentrated, particularly around migration bottlenecks such as adult salmon runs into rivers (Thompson et al., 2019). This focus on high-value prey during specific periods contributes to their observed impact on commercially important fish stocks. The main impact, however, is on coastal fisheries and aquaculture cage systems. Here they cause considerable damage to fish caught in fishing nets or damage aquaculture netting to access the fish, resulting in direct loss of or injury to stock and escape of stock through damaged nets. Requiring food intake between 4 and 6% of their body weight daily, seals can exert considerable pressure on fish stocks, especially those depleted by environmental degradation and fishing pressure. Their increasing presence and abundance is, however, introducing the concept of the predator pit, where the prey population is unable to recover after improved management of the stocks, reduction in fishing pressure or habitat restoration measures have been implemented.

One of the key distinctions between these mammalian predators lies in the scale and pattern of their predation. Otters exert chronic, low-to-moderate predation pressure across freshwater ecosystems. Their movements are relatively predictable, and their impacts are distributed over large areas, except when they target intensively stocked fisheries and aquaculture ponds. Seals, by contrast, exhibit highly concentrated predation, often focused on specific locations and times, such as salmon migration periods. Telemetry studies suggest that individual seals may repeatedly exploit particular rivers or structures, leading to localised hotspots of predation (Thompson et al., 2019). American mink are capable of exerting substantial predation pressure on freshwater fish, but have a diverse diet and pressure on fish stocks is variable

3.4 Ecological impacts of piscivorous fish

Fish themselves constitute a major group of predators of inland fish in Europe. Key piscivorous species include northern pike (*Esox lucius*), perch (*Perca fluviatilis*), pikeperch (*Sander lucioperca*), European (Wels) catfish (*Silurus glanis*), and brown trout (*Salmo trutta*) (Annexes 8.14-8.17). These species often function as apex predators, exerting considerable control over fish community structure (Craig, 1987, 1996; Hickley, 1986). Of these species, only brown trout is native across the majority of Europe (Wheeler, 1969), but the others have now been introduced in many water bodies outside their native range, often to enhance angling opportunities.

Piscivorous fish influence prey populations not only through direct consumption but also by inducing behavioural changes. For example, prey species may alter habitat use to avoid predation risk, which can affect growth and reproduction (Wootton et al., 1997). Seasonal and spatial variability in predation has also been documented, with factors such as temperature and habitat complexity influencing predator activity and success (Craig, 1987, 1996; Hickley, 1986).

Typically, piscivorous fishes regulate prey populations with numbers oscillating annually (Wootton et al., 1997), unless disrupted by habitat or environmental degradation (e.g. eutrophication or loss of vegetation), stocking of predatory fish species, or the introduction of non-native fish species, such as pike, pikeperch or Wels catfish. These conditions can cause destabilization of the predator-prey relationships and cascading effects that disrupt the natural balance. Typical examples are when pikeperch are introduced into slow flowing rivers or canals and predate heavily on small cyprinid populations (e.g. Hickley, 1986; Smith et al., 1996) or Wels catfish, a voracious predator, decimating native species populations, such as in the Ebro River in Spain (Fernández-Delgado and Herrera, 2026). Whilst these species may contribute valuable fisheries in their own right, they also degrade native wild fisheries and biodiversity, and indirectly alter ecosystem functioning, resulting in, for example, eutrophication (Mehner et al., 2004). It should be recognised these cases are not common; and introduction of new species is strictly regulated by national and European legislation (EU Regulation 1143/2014 on invasive alien species). They also rarely affect commercial ventures such as aquaculture operations or put-and-take fisheries, which are actively managed, and in some countries these species are farmed or used for targeted fisheries in enclosed water bodies (Cowx, 2015). It is when these species escape and colonise new bodies that problems arise, as seen from the Wels catfish and zander examples above, where considerable impact on native fish biodiversity is found (Hickley and Chare, 2004).

4 Economic impact of predators on freshwater fish and fisheries and aquaculture in Europe

Predation on inland fish in Europe is not only an ecological issue but also of considerable economic concern, particularly where wild fisheries, recreational angling, and aquaculture intersect with expanding predator populations. Information on the economic impact of fish eating predators on inland fisheries and aquaculture in Europe is, however, limited and largely restricted to the primary study carried out by EIFAAC/FEAP/EEA (FAO, 2026b) on fish eating birds as part of preparation for the drafting the Framework for a European Management Plan for the Great Cormorant (FAO, 2026a). The results of economic impact of cormorant depredation is reported by EIFAAC/FEAP/EEA (FAO, 2026b) and the results are synthesised here alongside complimentary information collated for other bird species and otter. The reader is referred to FAO (2026b) for details of the questionnaire and methodology for assessing the economic costs of predation on inland fisheries and aquaculture.

Across predator groups, economic impacts arise through direct fish losses, indirect effects on fish condition and behaviour, and the costs of mitigation and management. Note no information is available on the economic costs of seal predation on fisheries but the main impacts are reduced catches, including damage to sellable fish, and losses caused by predation on fish farms and damage to net enclosures and escape of stock. These all have direct implications on revenues from fish farming and livelihoods of fishers.

4.1 Direct economic losses on fisheries and aquaculture

The most accountable economic impact of predation is the direct removal of fish biomass, particularly in aquaculture systems and stocked fisheries. In pond aquaculture and open water fisheries, including stocked put-and-take fisheries, predators such as grey herons (*Ardea cinerea*) and cormorants (*P carbo carbo* and *P carbo sinensis*) can cause substantial losses because of high fish densities and limited refuges (Carss, 1993), coupled with naïve predatory responses of prey in production systems.

Studies on European fish farms report that bird predation can remove a significant proportion of stock, sometimes exceeding 20–30% annually in poorly protected systems (Carss, 1993). Otters can also cause damage in such settings, not only through consumption but via surplus killing, where multiple fish are injured or killed during a single foraging event (Kloskowski, 2005). Depredation of these levels makes the farming activities unsustainable and farms are closing under the pressure of uncontrollable predation levels (Zdeněk, et al. 2025). An example of the scale of depredation by birds is clearly shown in Figure 3. While this may be considered an extreme example, it is common across Europe where farmers are unable to protect their stocks and significant losses are incurred.



Figure 3. Fish farm in France illustrating the potential scale of bird depredation

A similar situation is found in intensively stocked fisheries, especially put-and-take and put-and-release fisheries. Here bird and otter predation can significantly compromise the fishery operation leading to loss of fisheries income and elevating costs of operation through continued need to stock.

In commercial river fisheries, particularly those targeting salmonids (*Salmo salar*, *Salmo trutta*), predators such as seals (*Phoca vitulina*, *Halichoerus grypus*), mergansers (*Mergus* spp), and great cormorants, are often implicated in reducing catchable stock. However, unlike aquaculture, fish losses here often relate to reduced catches as the fish grow and recruit to the exploitable fishery. Economic damage in these fisheries by predation is in addition to natural mortality from predation or other causes (Figure 2). The increased prevalence of predators following their rapid recovery over the past 20-30 years has seen fish population already stressed by other pressures unable to recover to ecologically sustainable levels.

Economic losses are particularly acute in artificial or highly managed systems, where ecological buffering capacity is low. In natural systems, predation is more diffuse and less directly translatable into financial loss. The somewhat intangible economic losses of biodiversity or failure to achieve good ecological status of water bodies are representative of economic losses.

Notwithstanding robust information is available on the direct economic losses caused by predation from the EIFAAC/FEAP/EAA 2024 survey (FAO, 2026b) beyond the economic impact of cormorants on fisheries and aquaculture. This is summarised in Table 3. The direct economic losses are considerable and, given they are not comprehensive surveys of all European countries, they highlight the growing reason behind calls for action to address the economic impact of predation on fish, fisheries and aquaculture enterprises in Europe.

Table 3. Regional overview of estimate annual damage by fish-eating predators on fish populations in reporting European countries (source FAO, 2026b)

Species	Number of countries reporting	Number of individual animals reported	Average occurrence in the year (days)	Estimated damages caused (Euro, average annual)	Reference years
Great cormorant	25	843,341	Variable between countries	671,874,150	Mainly 2023-2024
Pygmy cormorant	6	20,721	262	13,836,252	2023-2024
Grey heron	13	86,972	365	22,110,632	2023-2024
Great white egret	1	2,600	180	786,000	2023-2024
Great crested grebe	1	6,000	360	2,073,600	2024
Merganser	1	750	200	337,500	2024
Pelicans	1	14,200	90	7,033,000	2023-2024
Otter	8	13,343	355	25,168,025	2023-2024
Seals‡		3900		1,000,000	2009

‡ Source: Tuus - <https://www.nsrac.org/wp-content/uploads/2025/02/7HERKI1.pdf>

The economic damage and losses caused by cormorant predation to freshwater aquaculture in the EU alone are high, exceeding EUR 138 million annually, while total losses to aquaculture across Europe are estimated at around EUR 250 million. Pond fish farms are particularly vulnerable, commonly losing 10–30 percent of their fish stocks and, in extreme cases, up to 70 percent. Predation and the costs of scaring, guarding and netting are reducing farm profitability and have contributed to lower production and farm closures (Rey-Valette et al., 2024).

Cormorant predation also causes an estimated EUR 150–220 million loss in revenue in recreational and other inland fisheries from reduced catches and stock productivity, while costly stocking and habitat-restoration efforts undertaken by angling organizations and public authorities are undermined. Although evidence linking predator presence to reduced angling yields is often fragmented or indirect, predators such as seals, cormorants and mergansers can also reduce fish availability, particularly for high-value species like salmon. This has a repercussions for revenues

generated by angling through licences, tourism and associated spending (equipment, accommodation).

What is striking is the huge economic costs not only imposed by cormorants, but equivalent costs caused by otters, herons and other birds, despite the limited coverage. For example, economic losses reported for otters were in excess EUR 25 million from the 8 countries reporting, EUR 22 million for the 13 reporting heron damage and EUR 13 million for the 6 countries reporting damage from pygmy cormorants. If these results were extrapolated to cover all European countries, it is no surprise that conflict between fish-eating predators and fish and fisheries dominates the social and political agenda.

It is hard to account for the economic impact of introducing non-native fish species on native fish and ecosystems, but the greatest impact is likely the loss of fishing opportunities for native species. This, however, is compensated by the new fishing opportunities generated by the introduced species. The greatest impact will likely be the loss of endemic species biodiversity (Cowx 1998) and failure to meet WFD and HD ecological and conservation status targets. The cost of controlling non-native invasive species is also high (Diagne et al. 2021; Cuthbert et al. 2021) and should be considered in any economic assessment. Eradication of non-native species in large, open water bodies is also impractical, so control measures will incur recurrent costs.

4.2 Indirect economic effects

Indirect impacts are often overlooked but can be economically critical. Predator presence can alter fish behaviour, reducing feeding efficiency and growth rates (Kortan and Adamek 2011; Kortan et al. 2008, 2010), thereby lowering the market value of aquaculture products or the quality of fish available to anglers. Fish attacked but not consumed may suffer injuries (e.g. bite marks from birds or mammals), making them unsellable aquaculture products or less desirable for recreational catch. In recreational fisheries, stocking is often used to maintain fish populations. High predation rates can reduce the effectiveness of stocking programmes, increasing costs per surviving fish and lowering returns on investment. These effects are difficult to quantify but, in some cases, may exceed direct losses. This is particularly relevant in highly managed systems affected by cormorants, herons, egrets and otters.

A substantial proportion of the economic burden also arises from efforts to prevent or mitigate predation, including: installation of perimeter fencing and netting, use of deterrents (acoustic, visual, human presence), costs of guards and administrative and operational costs of licensing and control measures. Such interventions can be expensive and are not always effective, particularly as many predators, especially birds like cormorants, herons and egrets and otters rapidly habituate to such measures, reducing long-term effectiveness and requiring continual investment. Compensation schemes, where they exist, introduce additional complexity and do not eliminate underlying conflict.

Similar prevention measures are occasionally utilized in recreational fisheries bring additional costs to management and tens of thousands of volunteers from angling clubs spend many days every year to protect their fish from cormorant predation.

In some regions, limited culling of species such as cormorants or seals is permitted under licence. While these may reduce local predation, they incur administrative and monitoring costs, are often controversial and constrained by conservation law, and generally ineffective where predation is driven by mobile individuals, such as highly migratory birds or seals.

Some European countries provide compensation to fish farmers for predator-related losses. While this can offset some economic damage or costs for preventative measures, it introduces administrative burdens and challenges in verifying losses.

Conversely, healthy predator populations may also contribute to ecosystem services, including maintaining balanced fish communities, removing diseased or weak individuals and supporting biodiversity, particularly where they predate on invasive species, such as cormorants on round gobies in the Baltic area. These benefits are rarely monetised but may offset some negative economic impacts in the long term. It is when predator populations expand beyond the capacity of the water to accommodate predation and deplete the population or prevent recovery that problems become more evident and conflicts arise. Careful consideration must also be given when proposing fish-eating birds could act as a control mechanism for invasive species as this is counter intuitive of the EU Invasive Alien Species Regulation ([Regulation \(EU\) 1143/2014](#)) and the objectives of the EU biodiversity framework—the round goby is a non-native species of concern in Europe.

A critical issue is that the economic impacts of predation are unevenly distributed. Aquaculture operators and fishery managers bear the majority of direct costs whilst conservation organisations and the public benefit from predator protection and biodiversity. Anglers also experience losses, although these are often difficult to quantify. This asymmetry contributes to conflict, as those experiencing economic losses are often different from those benefiting from conservation outcomes.

5 Mitigation of impacts of predators on freshwater fish and fisheries and aquaculture in Europe

The impact of recovering fish-eating predatory species on inland fisheries and aquaculture is unequivocal and creating conflict between fisheries and aquaculture. This will likely escalate as bird and mammal populations continue to recovery under protective legislation, especially under the Habitats and Birds Directives. Various measures to prevent and avoid serious harm to wildlife and nature are available that can be applied under derogations issued according to Article 9 of the Birds Directive or Article 12 of the Habitats Directive. These measures have been reviewed and synthesized under the EU REDCAFE and INTERCAFE projects, and specifically within the Cormorant Toolbox (Russell et al., 2012) on the EU Cormorant Platform (EC, 2023b¹⁶) and more recently in advice offered by DG Environment¹⁷. The measures are designed and trialled for cormorants but the general consensus is that similar measures could be applied for other fish eating birds.

The main measures can be broken down as follows:

- Non-lethal measures:
 - scaring cormorants away from fishery or aquaculture units;
 - exclusion techniques;
 - habitat modification techniques to reduce the availability of fish to cormorants; and
 - fish stock management techniques to reduce the availability of fish to cormorants.
- Lethal measures:
 - lethal measures to reduce cormorant number directly; and

¹⁶ https://circabc.europa.eu/ui/group/e21159fc-a026-4045-a47f-9ff1a319e1c5/library/70d1e6e6-0a86-4498-8e66-f30819c374fe?p=1&n=10&sort=modified_DESC

¹⁷ https://environment.ec.europa.eu/publications/guidance-document-general-system-protection-bird-species-article-5-and-article-9-birds-directive_en

- reducing reproductive success through egg destruction.

Scaring is widely used across Europe to reduce cormorant, heron and egret predation, employing a range of visual and auditory techniques, including shooting, gas cannons, fireworks, lasers, reflectors and human presence. Emerging technologies, including automated detection using artificial intelligence, drones and more targeted deterrence systems, may improve the efficiency and precision of scaring and expand the management toolbox. Effectiveness depends on sustained, varied and coordinated deployment, and because it generally displaces rather than reduces birds, it may simply redistribute predation pressure among fisheries and fish farms. Consequently, scaring alone is unlikely to provide an effective solution. Its value is greatest when integrated with exclusion measures, habitat modification and appropriate fish-stocking practices. Wires, netting and increased habitat complexity can provide effective protection in small, enclosed or artificial systems, while stocking larger fish and avoiding periods of peak cormorant abundance can further reduce vulnerability. These approaches, however, become increasingly impractical in large, open waterbodies, where management should focus on site-specific combinations of measures and emerging technologies. Scaring and exclusion methods are also impractical for mergansers, grebes and pelicans, which are mostly found on open waters and rivers, and would disturb other wildlife. Regardless, scaring and exclusion methods have proved largely ineffective in a European context (FAO 2024), being restrictive, expensive to operate and birds become accustomed to the action.

Habitat modification, such as altering depth of ponds, increasing cover and providing refuge area are perhaps the best long-term mechanisms to alleviate bird depredation, but they are expensive to implement, change the behaviour of fish and are not always practical in recreational fisheries or aquaculture environments. Modification of stocking is already carried out, mostly involving stocking larger fish, changing the timing of stocking (at night and when fewer birds are around), and using scatter stocking to disperse the concentration of fish at the time of stocking. These all bring additional costs to fisheries management.

Lethal control of cormorants includes killing, egg oiling, egg pricking and destruction of nests, and in some cases felling trees used for roosting. These measures have had limited success in reducing either local or the wider European conflicts to address the proliferation of cormorant, heron or egret populations. Part of the problem is the differences in national implementation of derogations, including countries where lethal control is not permitted or is applied at insufficient intensity. This allows populations to be replenished through movement from areas with little or no control, especially for highly migratory species like cormorant and heron. Lethal control and reproductive interventions are unlikely to provide an effective solution, unless applied in a coordinated manner across the distribution range of the species at the pan European scale. Nevertheless, these interventions may contribute to managing localised conflicts when applied strategically alongside other measures.

These lethal and non-lethal control measures are widely used throughout Europe, as can be seen by the number of derogations issued between 2020 and 2024 under Article 9 of the Birds Directive since 2020 (Figure 4), but specifically *P. carbo carbo*, *Ardea cinerea* and *P. carbo sinensis*. Over 4000 derogations have been issued since 2020, with the most applications from Poland, France, Czech Republic, Hungary, Germany and Sweden. The numbers would be considerably greater if non-European Union countries were included, and indeed the UK had three times more Article 9 derogations for *P. carbo carbo* (2472 derogations between 2015 and 2019) than any European country (prior to leaving the EU in January 2020). Note Poland was the second highest for this period with 744 derogations. It should also be noted that many countries do not grant Annex 9 derogations or have not reported numbers awarded so the scale will be considerably greater. Also these are numbers of derogations issued and do not indicate the total number of birds killed or disturbed. Of further note

is a high proportion of the derogations were issued to control the target populations through killing and disturbance measures to prevent serious damage or protection of flora and fauna. These are uncoordinated and usually site specific, and fail to ultimately address the root cause of the problem.

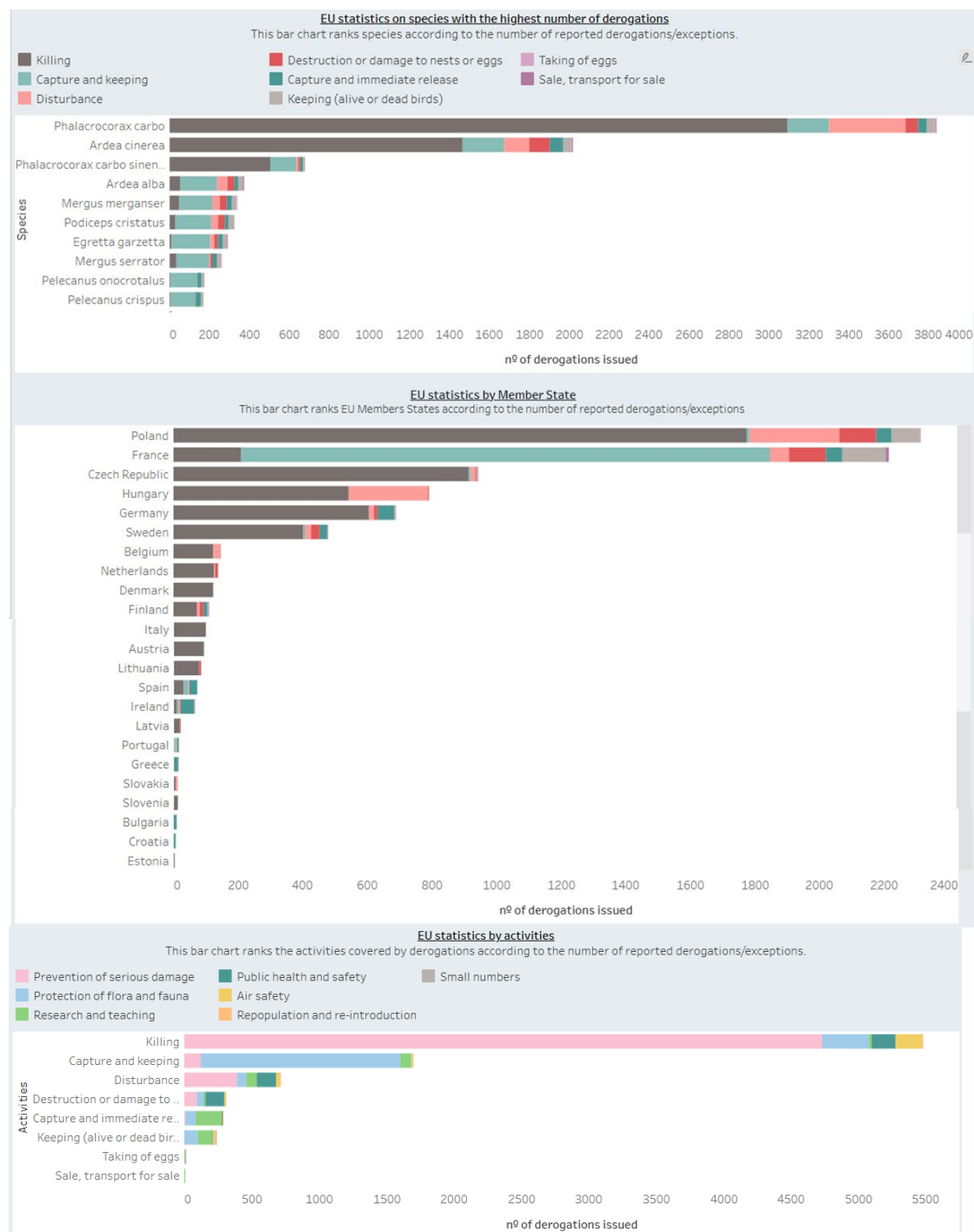


Figure 4. Number of derogations issued under Article 9 of the Birds Directive to control piscivorous in Europe: 2020–2025. Source: European Environment Agency (2024a) Overview of derogations and exceptions to species protection across the EU. <https://www.eea.europa.eu/en/analysis/maps-and-charts/overview-of-derogations-and-exceptions-dashboards>. Cited 07 August 2026

Complimentary measures are also applied to control fish-eating mammals but orientate around scaring, exclusion, capture and relocation, and in the last resort culling. Like birds, mammals become habituated to scaring tools and thus they have to be varied and continuous, often disturbing other

wildlife. Exclusion usually involves fencing to isolate the fish stocks from the predator. This requires continued maintenance and comes at high cost to isolate large water bodies or pond systems. Otters in particular find alternative routes of access. Acoustic scaring methods are also suggested for seals (Lehtonen et al. 2023; Veneranta et al., 2024), but mostly to keep seals out of fishing gears or those exploiting more confined river fisheries. Conversely, there is now considerable work to reduce bycatch of birds and mammals in fishing gears using exclusion devices or deterrents showing the fishing sector is cognisant of the potential impact it can have on protected predatory species (e.g. FAO, 2019; Lehtonen and Suuronen, 2025).

A similar situation, but on a lesser scale, exists for derogations under Article 12 of the Habitats Directive to manage mammal populations (Figure 5). Here nearly 400 derogations have been issued for mostly controlling otter populations, with most awarded to Poland, France, Ireland and Sweden. In the latter case, the derogations were also issued to control seal populations.

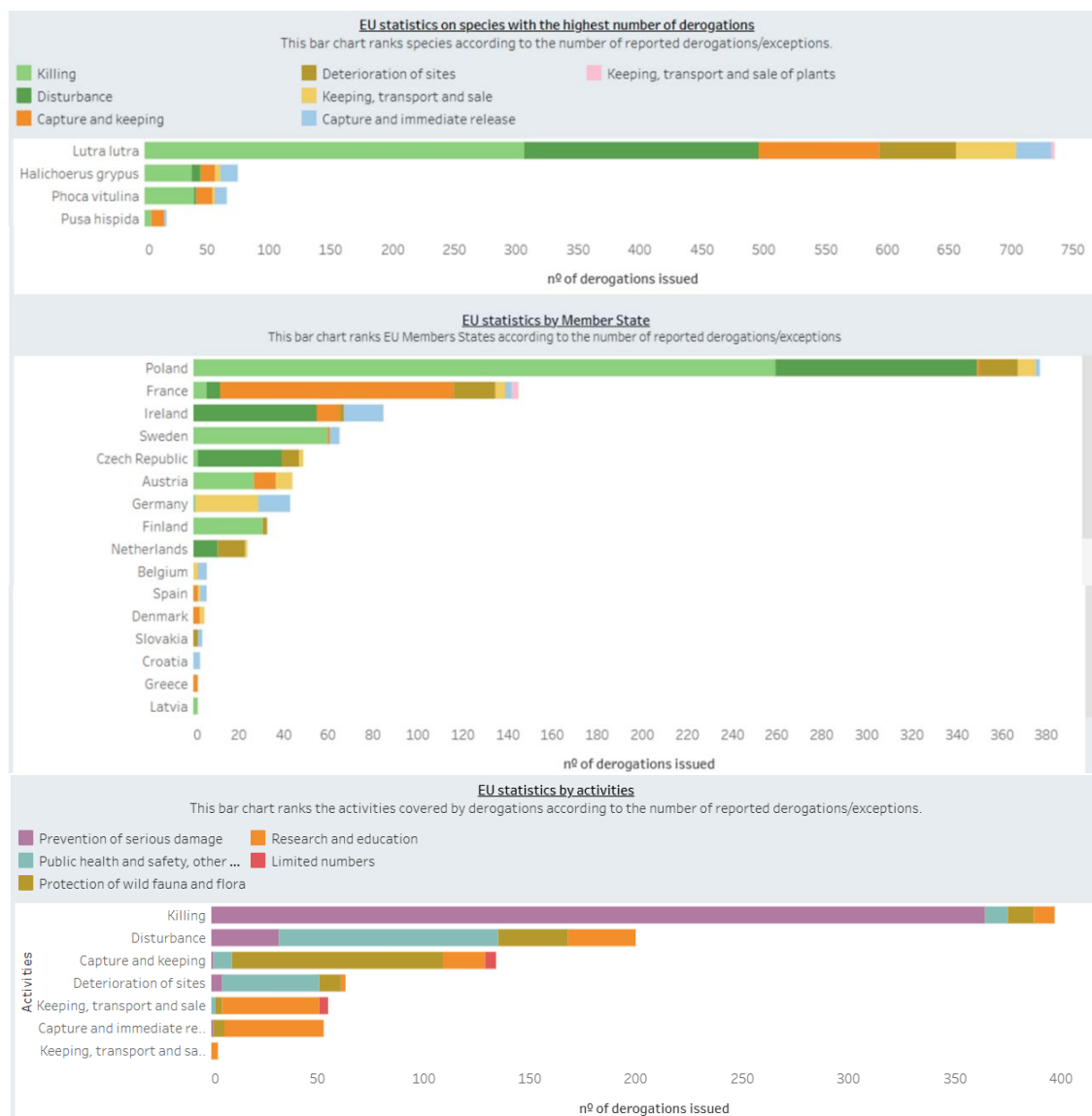


Figure 5. Number of derogations issued under Article 12 of the Habitats Directive to control piscivorous mammals in Europe: 2020–2025. Source: European Environment Agency (2024a) Overview of derogations and exceptions to species protection across the EU. <https://www.eea.europa.eu/en/analysis/maps-and-charts/overview-of-derogations-and-exceptions-dashboards>. Cited 07 August 2026

Despite the number of derogations to manage bird and mammal populations, questions must be asked about their effectiveness both in the short and long-term. For example, various measures are deployed to control great cormorants in different European countries to different extents, and with varying degrees of success (Russell et al., 1996, 2003, 2008; Russell and Carss, 2022). The choice of measures depends on the scale of the cormorant–fisheries conflict, the type of waterbody or fishery operation impacted, and the potential economic losses incurred. However, responses from a questionnaire on the cormorant-fish conflict circulated to EIFAAC member countries indicated these measures have met with varying, but usually limited, success (FAO, 2024). The main conclusion is that no single management intervention is effective at mitigating the problems created by great cormorants. Shooting (on a large scale) does not appear to be a viable option unless the numbers are reduced across the entire European distribution range. Continuous dispersal and turnover of birds is a result of incoherent action from countries that do not adopt intervention measures. Controlling local bird population density by destroying nesting areas and oiling eggs is, again, only likely to have limited and short-term impact, if not carried out in a coordinated manner at a regional scale, notably by targeting the main breeding colonies in northern Europe. Similarly, scaring methods (human disturbance, laser guns, and sound and taste aversion) do not appear to be effective because they must be carried out on a continuous basis. Moreover, birds become accustomed to the methods employed, and the problem is potentially passed on to other fisheries. Exclusion devices are only viable on some aquaculture facilities and are not feasible in open fisheries because they restrict or prohibit fishing activities.

Lessons can be learnt from the outcomes of attempt to control cormorant number, not least the need for coordinated, continuous interventions and the need for regional or pan-Europe plans for migratory species. While mitigation measures may control problem at a local level they come at considerable cost to the owner/operator often testing the economic viability of the fish farm or fishery. One option to offset these costs is financial compensation for losses from predation. A few countries provide compensation to fish farmers, but compensation is limited and rarely supports the heavy financial losses incurred. No compensation is available to support lost fisheries or fishing opportunities although a few countries offered limited compensation in the past to commercial (inland) fisheries. A few angling clubs involved in aquatic biodiversity protection have received financial support for preventive measures such as netting and construction of fish forests/shelters.

6 Information gaps and research needs

The recovery of bird and mammal populations across Europe represents a major conservation success. Legal protection under instruments such as the EU Birds and Habitats Directives has facilitated population growth and habitat recolonization. However, this recovery has led to increased conflict with fisheries, particularly in regions where aquaculture and fisheries are economically important and where predation can result in financial losses. There is now a wealth of information and supporting data to confirm that predation on inland and coastal fisheries and aquaculture in Europe is extensive and having considerable ecological and economic impacts on fish biodiversity, fisheries and aquaculture in inland waters. Yet there remains scepticism over the precarious state of inland fisheries and aquaculture and the contribution of fish-eating predators to the evolving deterioration of these sectors or failure of fish populations to recover from extensive habitat improvement measures.

The reasoning is complex and multifaceted involving a diverse array of taxa. It is not disputed that capture fisheries have consequences on fish populations, but many fisheries have now ceased to

operate, e.g. pound fisheries in Denmark or coastal net fisheries in UK, and the fish stocks have not recovered. This is intrinsically linked to the state of the waters, which are in a precarious state (WFD status: European Commission, 2025. European Environment Agency, 2024.), and affects not just fish species but aquatic biodiversity in general, bring recovering species into conflict for depleted resources. Further, piscivorous birds, mammals and even fish each play distinct roles in shaping freshwater ecosystems, with interactions occurring across multiple spatial and temporal scales. They typically function as resident or transitional apex or mesopredators, exerting continuous top-down control within freshwater systems. Their impacts are embedded within ecosystem processes and are often stabilising. It is the rapid expansion and increase in abundance of predators that has altered the ecological balance or created conflict with the fisheries and aquaculture sectors, notable through impact on commercial and recreational ventures. There is also a mismatch between conservation of apex, highly visible, predatory species and fish species that are considered a commodity rather than accounting for their intrinsic biodiversity value.

The limited success of management measures recommended by the European Commission and others (see EU Cormorant Platform (EC, 2023b¹⁸) and more recently in advice offered by Directorate General Environment¹⁹, has led to the conclusion that a pan-European management plan for at least one species, the great cormorant, is needed (FAO 2026a). A similar conclusion to prepare an African-Eurasian Waterbird Agreement (AEWA) single species management plan for the Dalmatian pelican had previously been recommended but the plan has not been implemented (Catsadorakis and Portolou 2018). Whilst a pan European or AEWA plan is probably not necessary for other species because the impact is more localised, affirmative actions are needed to protect freshwater biodiversity, inland fish and fisheries and aquaculture enterprises. Solutions are not mass culling as falsely represented by antagonists of management of bird fish conflicts²⁰, but strategic, coordinated, adaptive management of fish and fisheries, birds and mammals and the ecosystems and habitats on which they rely is needed, as proposed under the Framework for a European Management Plan for the Great Cormorant (FAO 2026a). As part of this process, considerations need to be given to the critical role fish and fisheries play in terms of contribution to biodiversity, protection of habitat and ecosystem functioning, and the services they deliver.

Effective management will require an integrated approach that considers ecological functioning, economic costs, and social contexts. Moving the dialogue away from single species protection management towards ecosystem and wider biodiversity management frameworks will help answer Carss's question in the title of his thoughtful paper (Carss, 2021) *"There must be some kind of way out of here"*. Carss argues that "Much of the controversy over [Cormorants] is fuelled by differences of opinion and, coupled with its persistence and entrenched nature, it has many of the characteristics of a so-called intractable environmental conflict". Future research should prioritise long-term, interdisciplinary studies that bridge ecological and economic perspectives, enabling more balanced and evidence-based decision-making where all components of biodiversity, economic development and livelihoods are considered, i.e. addressing uncertainty and conflicting values (Termeer et al., 2019).

¹⁸ https://circabc.europa.eu/ui/group/e21159fc-a026-4045-a47f-9ff1a319e1c5/library/70d1e6e6-0a86-4498-8e66-f30819c374fe?p=1&n=10&sort=modified_DESC

¹⁹ https://environment.ec.europa.eu/publications/guidance-document-general-system-protection-bird-species-article-5-and-article-9-birds-directive_en

²⁰ <https://www.birdlife.org/news/2025/09/23/mass-culling-of-cormorants-wont-save-fish/>

More clarity is also needed regarding international management frameworks to resolve conflicts, both locally and regionally. It is unsurprising that local actions are not wholly effective given several predatory species, especially cormorants and herons, are highly mobile and replenish populations from external sources where management control measures are not carried out. Consequently different packages of actions are needed in different areas but applied in a regionally coordinated manner, similar to that proposed for the great cormorant (FAO, 2026a). Given the structural features of human-wildlife conflicts, purely control-oriented approaches are unlikely to produce long-term resolutions. Instead, adaptive governance strategies that emphasize monitoring, local knowledge and feedback, and stakeholder inclusion should enhance legitimacy and effectiveness.

In addition, despite extensive research, several issues remain unresolved. One key debate concerns the magnitude of predator impacts on fish populations, particularly in relation to commercially valuable species and aquaculture. While studies suggest variable long-term effects, disagreements persist, often influenced by stakeholder perspectives. One key issue is the scaling of impacts, as most studies are conducted at local scales and may not capture broader population-level effects. There is a need for long-term, large-scale studies that integrate ecological and environmental variables. In addition, there is need for more robust and up to date data on trends in predator and prey population abundance and distribution, and economic impact assessments.

This discrepancy between regional ecological assessments and local economic experiences contributes to ongoing conflicts between conservationists and fisheries stakeholders. Effective management requires a balanced approach that considers both the conservation of biodiversity and economic realities of conflicts arising. Potential strategies include improved fishery design, targeted exclusion measures, and compensation schemes for affected operators. This will require specific action plans to guide management of the species in accordance with the Birds Directive and Habitats Directive. These action plans are designed to resolve local problems through coordinate management that engages all sectors and all aspects of biodiversity conservation, including maintaining and enhancing economic sectors.

Another issue relates to the cumulative effects of multiple predators. Predators coexist with other piscivores, yet few studies have examined their combined impacts on fish communities. Understanding these multi-predator systems is essential for developing comprehensive models of freshwater food webs. Additionally, climate change and habitat alteration may modify predator–prey dynamics, necessitating further research. Ultimately, resolution of the conflict will require better collaboration between groups.

7 Conclusions and recommendations

7.1 Conclusions

1. **Predation is a significant pressure on European freshwater fish resources, fisheries and aquaculture.** Its magnitude varies considerably among predators, locations and production systems. Predation is a natural ecological process, but recovering populations of several fish-eating birds and mammals can exert substantial additional pressure where fish populations are already depleted, concentrated during migration, or confined in aquaculture and managed fisheries. Predation impacts should therefore be considered alongside habitat degradation, river fragmentation, pollution, invasive species, climate change and other pressures when assessing the status and recovery prospects of fish populations.

2. **Successful recovery of protected predators can create conflicts with the conservation of freshwater fish and wider aquatic biodiversity.** European conservation measures have contributed to major recoveries of cormorants, otters and other predators. However, conservation success for one group of species should not inadvertently undermine conservation objectives for other components of biodiversity. Many European freshwater fish populations are already in poor condition and some vulnerable populations may be unable to recover where high predation pressure coincides with depleted fish populations, disrupted reproduction and recruitment within the population, and other stressors.
3. **The ecological effects of predation are highly context-specific, but can be severe at vulnerable locations and life stages.** Particularly important situations include spawning and migration bottlenecks, salmonid smolt migrations, depleted river fish populations, small and shallow water bodies, stocked fisheries and aquaculture ponds. Highly mobile predators, particularly cormorants and herons, can exert pressure across national boundaries, while otters, mergansers, pelicans and seals can cause substantial localized impacts. Introduced predatory fish can also adversely affect native fish communities and ecosystem functioning.
4. **Predation causes substantial economic losses to European aquaculture and fisheries.** The evidence reviewed indicates annual losses exceeding EUR 138 million in EU freshwater aquaculture and around EUR 250 million across European aquaculture, while cormorant-related losses to recreational and other inland fisheries are estimated at EUR 150–220 million annually. Reported losses from otters, herons and other predators add tens of millions of euros, despite incomplete geographical coverage. These figures demonstrate that predator–fish conflicts are not solely conservation issues but also economic and rural-development concerns.
5. **The real economic burden of predation is considerably broader than the value of fish consumed alone.** Fish farmers and fishery managers also incur costs from injured fish, reduced growth, repeated stocking, damaged infrastructure, fencing and netting, guarding, scaring, administrative procedures and other mitigation measures. Recreational fisheries can experience reduced catches and participation, while local economies can lose angling-related expenditure and tourism. Predation can also undermine public and private investments in fish stocking, habitat restoration and recovery programmes.
6. **Current mitigation measures are generally insufficient to resolve persistent predator–fish conflicts.** Scaring, exclusion, habitat modification and locally applied lethal control can reduce losses, especially in small or enclosed systems, but their effectiveness is often temporary and implementation costly. Mobile predators can move between sites and across national borders, effectively transferring rather than resolving the problem. Thousands of derogations have been issued under European nature legislation, but their fragmented implementation demonstrates the limitations of predominantly local approaches for widespread or migratory predator populations.
7. **The costs and benefits of predator conservation are unevenly distributed across society.** The wider public benefits from the recovery and protection of wildlife, while fish farmers, fishery managers, angling organizations and other rural enterprises frequently bear a disproportionate share of the resulting costs. Compensation for verified losses and preventive measures exists only in some countries and varies considerably. Without mechanisms for sharing these costs, predator conservation policies risk reducing the economic viability of aquaculture and fisheries, and increasing conflict between conservation and fisheries stakeholders.
8. **Europe lacks a sufficiently coordinated policy and institutional framework for balancing predator conservation with fish conservation and sustainable fisheries and aquaculture.**

Existing arrangements remain fragmented across nature conservation, water, fisheries and aquaculture authorities, and particularly across national borders. The problem is particularly evident for highly mobile species such as the great cormorant and grey heron. A more balanced approach is required in which fish biodiversity, fisheries, aquaculture, protected predators, ecosystem functioning and socio-economic interests are addressed within the same decision-making framework.

7.2 Recommendations

1. Integrate predation explicitly into European and national aquatic biodiversity and fisheries policies.

The European Commission and EIFAAC Member countries should recognize significant predator pressure as one of the factors that may affect fish population status, recovery and achievement of aquatic biodiversity objectives. Where evidence demonstrates significant impacts, predation should be considered alongside habitat degradation, barriers, water quality, invasive species, climate change and exploitation in assessments and recovery programmes. Implementation of the Water Framework Directive, Habitats Directive, Birds Directive and relevant fisheries and aquaculture policies should seek complementary outcomes for predator conservation, freshwater fish biodiversity and sustainable fisheries, rather than addressing these objectives through separate sectoral processes.

2. Review implementation of the Birds and Habitats Directives to enable timely, proportionate and coordinated responses to serious damage.

The European Commission and EIFAAC Member countries should review how derogations relating to fish-eating predators are implemented, with the aim of reducing unnecessary administrative complexity and improving consistency among countries. Clear guidance should define when serious ecological or economic damage justifies intervention and the evidence required. Where populations are favourable and damage is demonstrated, authorities should be able to apply proportionate management measures without excessive delays. For highly mobile species, national derogations should be embedded within coordinated regional or European management frameworks rather than implemented as isolated local actions.

3. Establish coordinated management frameworks for predator species according to the geographical scale of their impacts.

For highly mobile species whose impacts cross national borders, particularly the great cormorant, heron and seals, European countries should implement coordinated population-level management. The EIFAAC Framework for a European Management Plan for the Great Cormorant (FAO 2026a) provides an appropriate basis for such action. For predators whose impacts are predominantly national or local, governments should prepare species or conflict-management plans identifying priority areas, intervention thresholds, responsible authorities and permitted measures. Plans should combine prevention, habitat measures, targeted population management where justified, and protection of vulnerable fish populations and aquaculture facilities.

4. Establish fair financial mechanisms to compensate and support sectors bearing predator-related costs.

European and national authorities in EIFAAC Member countries should develop transparent mechanisms to support fish farmers, fishery managers and organizations that experience demonstrable serious damage from protected predators. Support should cover not only verified fish losses but, where appropriate, reasonable investments in effective preventive infrastructure and measures. Funding mechanisms should recognize that wildlife conservation

provides public benefits while its costs may fall disproportionately on individual enterprises and fishery organizations. Compensation should complement rather than replace predator management and should be linked to reasonable preventive measures and standardized damage-assessment procedures.

5. Strengthen institutional coordination and give fisheries and aquaculture authorities an equal role in predator-management decisions.

Governments should establish permanent coordination mechanisms involving authorities responsible for environment and nature conservation, inland and coastal fisheries, aquaculture and water management, together with research institutions and relevant stakeholder organizations. At the European level, stronger cooperation is needed between institutions responsible for biodiversity, water, fisheries and aquaculture policies. Decisions on predator management should consider ecological, economic and social evidence together and explicitly assess consequences for protected and threatened fish populations, aquaculture viability, recreational and commercial fisheries, ecosystem restoration investments and predator conservation.

6. Establish a European monitoring and evidence system focused on management decisions and measurable outcomes.

European countries, supported by EIFAAC and relevant European institutions, should harmonize monitoring of predator abundance, fish population status, aquaculture and fishery losses, mitigation expenditure and management interventions. Research institutions should prioritize major evidence gaps that directly affect management decisions, including predator impacts on vulnerable fish populations, cumulative impacts of multiple predators and the effectiveness and cost-efficiency of interventions. Monitoring should be linked to agreed indicators and periodic evaluation so that ineffective measures can be discontinued and successful approaches expanded. Research should support action rather than become a prerequisite that indefinitely delays proportionate management where serious damage is already adequately demonstrated.

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9 Annexes

9.1 Great cormorant - *Phalacrocorax carbo carbo* and *Phalacrocorax carbo sinensis*

9.1.1 Distribution and abundance

The great cormorant comprises two sub-species, *Phalacrocorax carbo carbo* often termed the Atlantic subspecies and *Phalacrocorax carbo sinensis* the continental cormorant (Figure 6). Both sub-species have widespread distribution and have overlapping ranges in the North Sea area. They are progressively expanding their ranges south and west, especially *P. carbo sinensis*. *Phalacrocorax carbo carbo* mainly breeds and inhabits north-west Europe and the west coast of Scandinavia as far south as Spain. There are potentially two breeding populations of *P. carbo sinensis*, one in north and central Europe and a second around the Black Sea and Mediterranean. There is also another population around the Caspian Sea and south-west Asia. The European population is estimated at 401 000-512 000 pairs, which equates to 803 000-1 020 000 mature individuals (BirdLife International 2015) (BirdLife International 2021; <https://datazone.birdlife.org/search>). These data are outdated, relating to a European count in 2013 before expansion in abundance and range of the species. Various estimates put the numbers in the range 1.5-2 million in Europe (see FAO 2024, 2026a). The species is listed as Least Concern on the IUCN Red List and the overall population trend in Europe is considered to be increasing (BirdLife International 2021).

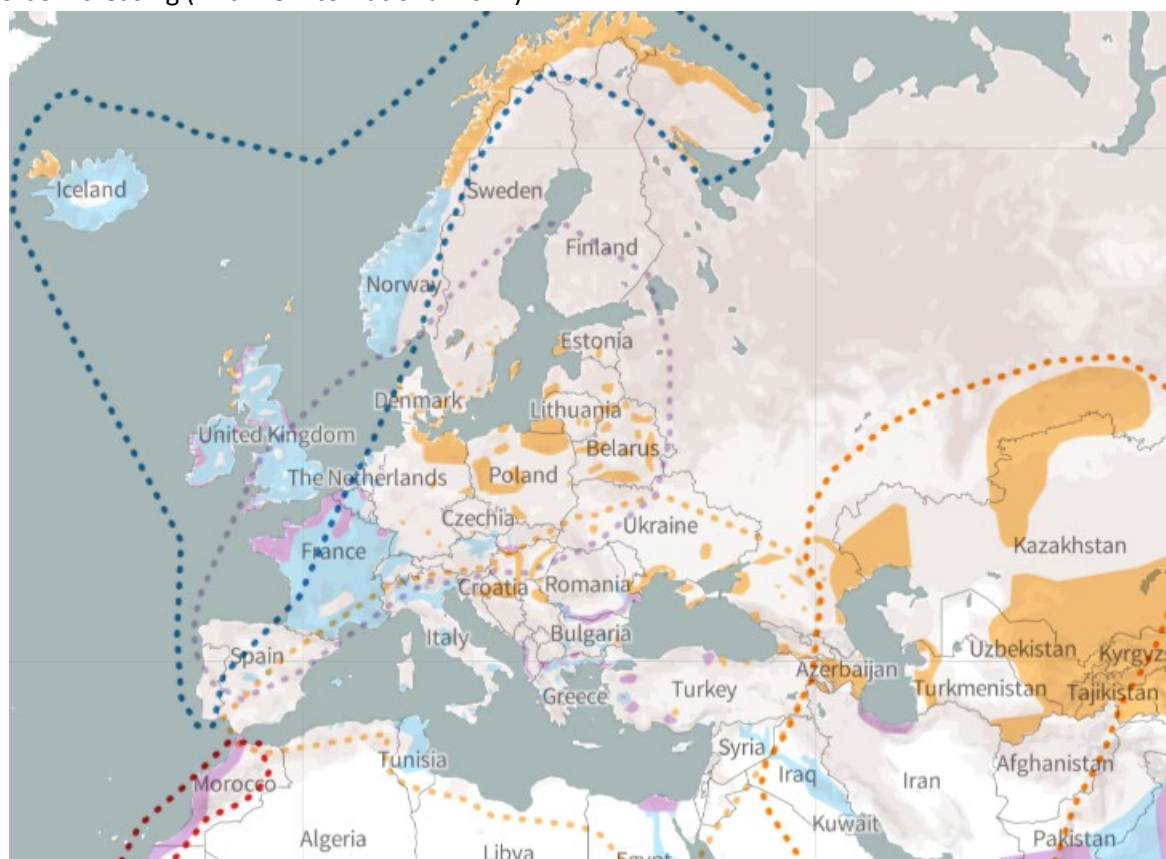


Figure 6. Distribution of great cormorant in Europe highlighting location of resident ●, breeding ● and non-breeding ● areas and discrimination of discrete populations (dotted line) (Source: <https://criticalsites.wetlands.org/en>)

9.1.2 Habitat and ecology

Cormorants are colonial waterbirds that breed in relatively large colonies. They are flexible with regards where they establish colonies, and build their nests in trees, shrubs and/or on the ground. They breed directly on the ground on small islands if these are safe against predators (primarily foxes). However, if there are trees and shrubs on the island where they settle, they usually choose to build their nests in them. When cormorants breed by lakes, the nests are often found in trees next to the lakeshore.

Cormorants have established breeding colonies in most European countries, but most of the breeding takes place in northern Europe, especially around the Baltic Sea (van Eerden et al., 2012). From late summer to autumn, there is a shift in the distribution of cormorants away from the Baltic/Nordic fjords and freshwater areas and out to the more open coasts and remote small islands. Around September–October they begin their autumn migration. Some migrate along the Atlantic coast and others migrate over land, usually along rivers (Figure 7) (Frederiksen, et al., 2018). Important wintering areas include France, southern Germany, northern Italy, The Kingdom of the Netherlands, Spain and Switzerland (Bregnballe and Rasmussen, 2000). Some cormorants choose to stay in northern areas, including the British Isles and the Baltic Sea region in winter, and do well in mild winters. The number of cormorants that overwinter in the north has increased as winters have become milder, linked to increasing air temperatures and less ice cover.

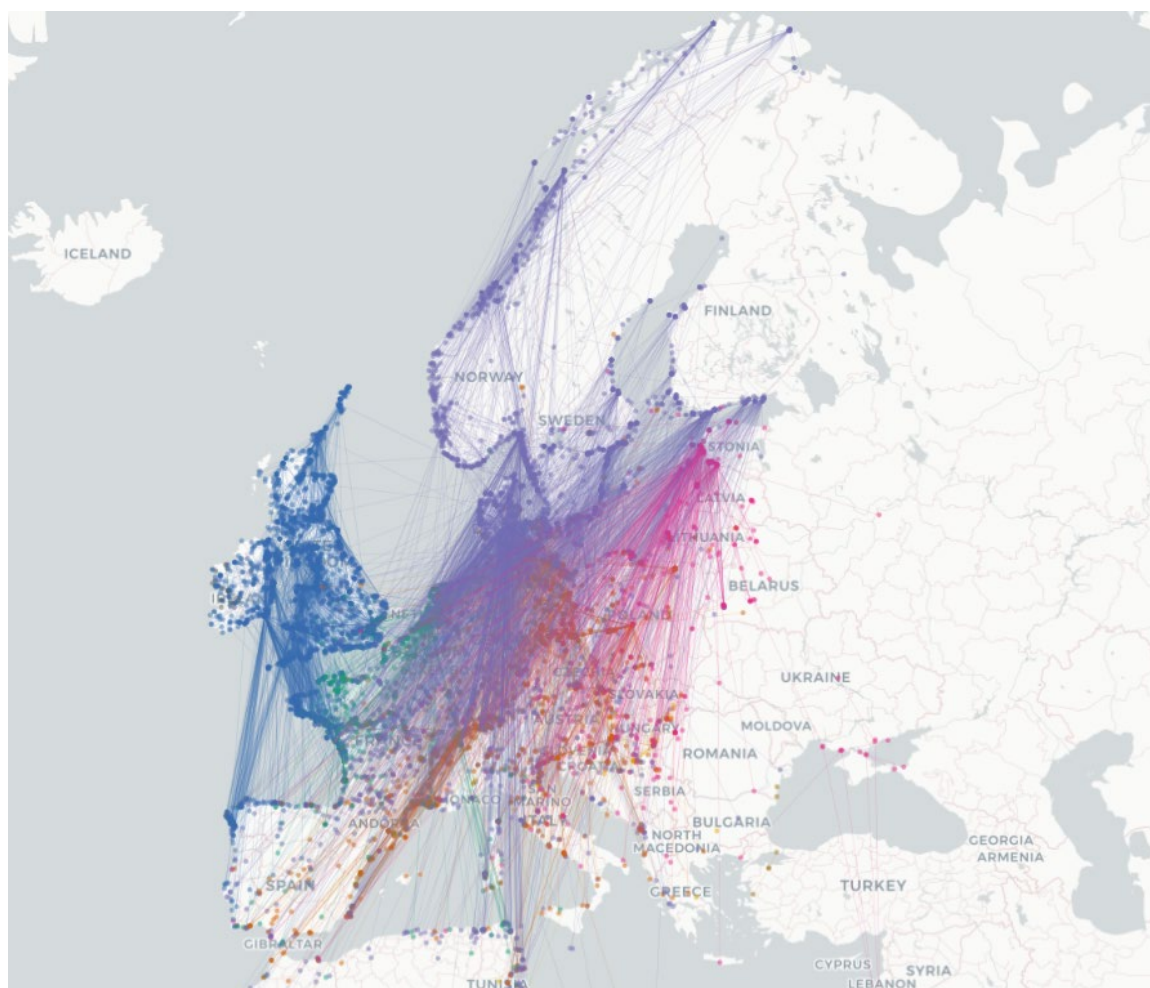


Figure 7. Migration patterns of great cormorants from The Eurasian African Bird Migration Atlas Source: Bird migration atlas: great cormorant - <https://migrationatlas.org/node/1773#section1>

Cormorant numbers in southern European countries like Croatia, Italy, Portugal and Spain have continued to increase in recent decades (Regione del Veneto, 2024; Opacak et al., 2004; Junta de Andalucía, 2025), as well as in Central and Eastern Europe (Bregnballe et al., 2014). Cormorants in Central and Eastern Europe tend to stay year-round, so they are moving from obligatory migratory birds towards more diverse strategies (including resident birds).

Cormorants live almost exclusively on fish. They are pursuit-diving predators, capable of capturing fish underwater with high efficiency. Unlike wading birds such as grey herons (*Ardea cinerea*), cormorants can exploit deeper and more structurally complex habitats, including lakes, reservoirs, and large rivers (Bregnballe, 2009). Cormorants usually seek food alone, but also forage in groups of hundreds in fjords, lakes, rivers and in shallow marine areas. Cormorants are good at locating areas with many fish that are relatively easy to catch, such as in ponds and small open lakes (van Eerden et al., 2012).

Common prey in inland waters includes cyprinids, percids, and salmonids (Carss, 2003). Although often portrayed as indiscriminate predators, studies indicate some degree of prey selectivity, influenced by factors such as fish size, behaviour, and habitat use. Cormorants typically target fish within a manageable size range (often 15–50 cm), balancing energetic gain with handling efficiency. Cormorants can survive on shrimps, sticklebacks and tiny sand goby if other prey is absent, but they can also eat fish of up to 2 kg ($\pm$ 50 cm in length) (Klenke et al., 2013; Kallö et al., 2023). A cormorant's food intake fluctuates throughout the season from 200 to 700 g/day, with a mean of 500 g/day (Grémillet et al., 1996; Keller and Visser, 1999; Ridgway, 2010).

9.1.3 Ecological and economic impact of predation

Discussions on cormorant predation on wild fish, and thus commercial fishing, recreational fishing, fisheries and fish conservation, have been intense for decades (Kindermann, 2008; Cowx, 2013; Carss, 2022; Ovegård et al. 2021; Saarikoski et al., 2025). Impacts from cormorant predation on wild fish populations are, however, difficult to measure. Consequently, most information is gained from single-site assessments carried out as part of targeted studies (see Kindermann, 2008; Cowx, 2013) and the EU Cormorant Platform (EC, 2023). These are supplemented by information in various national and regional cormorant management plans (e.g. Denmark, Finland, the Veneto region in Italy, and Sweden; see Gerdaux, 2005 and Cowx, 2013 for overviews) and information being compiled as part of the EU funded ProtectFish project and EU-funded Framework for a European Management Plan for the Great Cormorant (FAO, 2026a). These studies have been discussed in detail in the Framework for a Management Plan (FAO 2026a) and it is not the intention to repeat the output here. These studies, however, provide clear and compelling evidence for predation impact on fish species and populations in specific areas. There are many common recurrent results from different places that show the impact of cormorant predation on fish stocks, especially salmon and sea trout [smolts] and grayling in rivers, pike in lakes, and cod and juvenile flatfishes in coastal waters.

There is now substantial evidence to show that cormorant predation can have considerable adverse impacts on aquaculture and inland and coastal fishing, and on aquatic biodiversity in general (Ekblad et al., 2026). Considering a total population of 2 million cormorants in Europe (as a geographical area, rather than solely the European Union) and the fact that they each consume a mean of 500 g of fish/day (Grémillet et al., 1996, Keller and Visser, 1999, Ridgway, 2010), equates to $\approx$ 365 000 tonnes of fish consumed each year, assuming they all forage within European waters the whole year. When compared to commercial fishing landings from the high seas, this number is only a fraction of the total – but if it is compared with total combined annual coastal and freshwater fish harvest around Europe, it is a very high proportion. Cormorant predation has also contributed to collapse of cod stocks in coastal waters and specifically preventing recovery of stocks under stress (Jepsen et al. 2026).

This is further backed up by the EIFAAC/EAA/FEAP report (FAO 2026b) on the economic impact of cormorants where an estimated EUR 670 million was lost to cormorant predation in 25 reporting countries, either by direct consumption, damage and disturbance to fish and cost of mitigation and prevention measures (Table 4). No account is taken of the economic losses in terms of biodiversity or livelihoods. Many fish farms that operate on tight margins are closing, especially extensive carp farms in Eastern Europe (Zdeněk, et al., 2025). There is also no account taken of damage to trees, loss of aquatic fauna and flora in the vicinity of breeding and roosting sites (e.g. Goc et al., 2018), loss of other bird species (Meitern, et al., 2026) or the failure of recovery of water bodies to achieve good ecological status under the WFD.

Table 4. Regional overview of average damage by great cormorant on fish populations in European countries (source FAO, 2026b)

Country	Breeding cormorant population (individuals)	Non-breeding population (individuals)	Estimated damages caused (EUR, average annual)	Reference years	Notes / basis
Austria	3036	3997	3 362 700	2023-2024	Available data from Upper Austria/Styria plus Lake Constance; subnational totals aggregated.
Belgium	0	427	388 000	2023-2024	Wallonia fish farms; breeding population not reported.
Croatia	3300	20 000	8 985 000	2013-2018	National estimate based on EEA/Birds Directive reporting and consumption assumptions; reliability noted as low in summary.
Cyprus	750	5 125	2 718 000	2023-2024	Six fish farms; breeding and non-breeding damage combined.
Czechia	1028	14 898	6 745 700	2023-2024	Czech Fish Farmers Association and Czech Anglers Union data; 2023 and 2024 averaged.
Denmark	61 470	39 000	91 796 000	2023-2025	Breeding/non-breeding and damage estimates averaged across available 2023-2025 data.
Estonia	84 150	Unknown	44 178 500	2023-2024	National breeding estimate; non-breeding not reported.
Finland	59 870	29 935	40 412 250	2023-2024	Breeding and non-breeding damage combined.
France	22 269	117 533	34 796 000	2021/2024	2023 estimate uses 2021 counts; 2024 uses 2024 breeding and wintering counts.
Germany	46 747	36 179	25 670 500	2023-2024	Breeding and wintering damage combined.
Greece	20 000	41 750	29 950 000	2023-2024	Breeding and non-breeding damage combined.
Hungary	1 730	8 881	7 012 500	2023-2024	Breeding and non-breeding damage combined.

Latvia	5 740	Unknown	2 550 000	2023-2024	Breeding population only; non-breeding unknown.
Lithuania	16 700	Unknown	6 985 000	2023-2024	Breeding population only; non-breeding unknown.
Republic of Moldova	12 000	20 000	12 800 000	2023-2024	Breeding and non-breeding damage combined.
The Netherlands	37 000	41 000	28 470 000	2023-2024	2024 wintering range converted to midpoint for averaging.
Norway	43 400	28 379	46 440 000	2015/2020	Adult breeding and 1-/2-year old bird estimates; carbo and sinensis components combined.
Poland	58 166	39 003	42 520 000	2023-2024	National estimate; sedentary and fly-over non-breeding components combined.
Romania	28 500	77 778	44 720 000	2023-2024	Breeding and non-breeding damage combined.
Serbia	1 980	14 000	8 107 000	2023-2024	Non-breeding range converted to midpoint.
Slovakia	0	11 500	4 313 000	2023-2024	Average non-breeding damage.
Spain	1 500	75 000	35 119 000	2023-2024	Breeding and non-breeding damage combined.
Sweden	149 534	149 534	80 748 000	2023	Breeding and non-breeding estimates from 2023 summary table.
Switzerland	7 311	11 922	0	2023-2024	Damage reported as zero in summary table; 2024 non-breeding not available, so 2023 retained.
Ukraine	140 000	12 500	22 125 000	latest available	Single-year estimate from summary table.
UK	19 000	45 000	40 962 000	latest available	Single-year estimate from summary table.
Total of 25 countries	825 181	843 341	671 874 150		

9.1.4 Mitigation measures

Numerous reviews of measures to prevent and avoid serious harm by fish-eating birds to inland fisheries and aquaculture enterprises have been undertaken, most of which have been synthesized under the EU REDCAFE and INTERCAFE projects, and specifically within the Cormorant Toolbox (Russell et al., 2012). The main measures can be broken down as follows:

- Non-lethal measures:
 - scaring cormorants away from fishery or aquaculture units;
 - exclusion techniques;
 - habitat modification techniques to reduce the availability of fish to cormorants; and
 - fish stock management techniques to reduce the availability of fish to cormorants.
- Lethal measures:
 - lethal measures to reduce cormorant number directly; and
 - reducing reproductive success through egg destruction.

The extent of the problem with cormorant predation is exemplified by the number of derogations issued under Article 9 of the Birds Directive for both *P. carbo carbo* and *P. carbo sinensis*. More applications have been awarded for these two cormorant species than any other species, except grey heron (*Ardea cinerea*) since 2020 (Figure 4), and the numbers would be considerably greater if non-European Union countries were included. It should also be noted that many countries do not grant Annex 9 derogations or have not reported numbers granted so the scale will be considerably greater. Also these are numbers of derogations granted and do not indicate the total number of birds killed or disturbed. Of further note is that most of the derogations were for killing or disturbing birds to prevent serious damage or protection of flora and fauna.

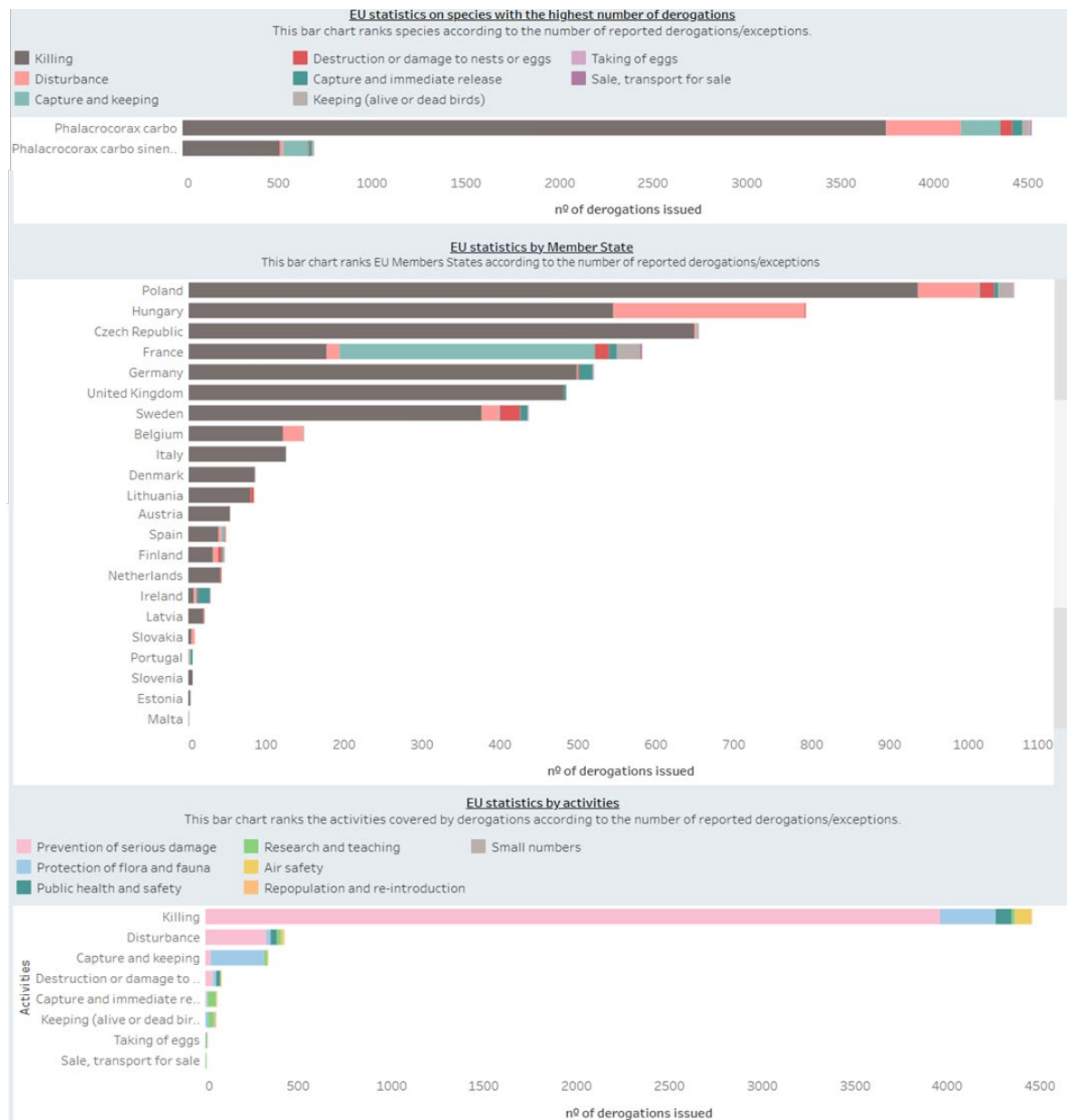


Figure 8. Number of derogations issued under Article 9 of the Birds Directive to control great cormorants in Europe: 2020–2025. Source: European Environment Agency (2024a) Overview of derogations and exceptions to species protection across the EU. <https://www.eea.europa.eu/en/analysis/maps-and-charts/overview-of-derogations-and-exceptions-dashboards>. Cited 07 August 2026

The responses from a questionnaire circulated to EIFAAC member countries indicate, however, that these measures have met with varying, but mostly limited, success (FAO, 2024). The main conclusion is that no single management intervention is effective at mitigating the problems created by great cormorants. Shooting (on a large scale) does not appear to be a viable option unless the numbers are reduced across the entire European distribution range. Continuous dispersal and turnover of birds as a result of incoherent action from countries that do not adopt intervention measures. Controlling local bird population density by destroying nesting areas and oiling eggs is, again, only likely to have a limited and short-term impact, if not carried out in a coordinated manner at a regional scale, notably by targeting the main breeding colonies in northern Europe. Similarly, scaring methods (human disturbance, laser guns, and sound and taste aversion) do not appear to be effective because they must be carried out on a continuous basis. Moreover, birds become accustomed to the methods employed, and the problem is potentially passed on to other fisheries or fish farms. Exclusion devices are only viable on some aquaculture facilities and are not feasible in open fisheries because they restrict or prohibit fishing activities. These outputs and others have led to conclusion that a pan-European management plan for the great cormorant is needed (FAO 2026a).

One other dimension that needs discussion relates to cormorants playing a role in controlling the invasion of round goby (*Neogobius melanostomus*) in certain parts of Europe, as it has become an important prey resource for cormorants²¹. Considering cormorant predation as a control mechanism is counter intuitive of the EU Invasive Alien Species Regulation ([Regulation \(EU\) 1143/2014](#)) and the objectives of the EU biodiversity framework– the round goby is a non-native species of concern in Europe. Its introduction has altered benthic fish communities, reduced native species distribution and abundance through competition and predation, and modified food webs across many river basins: these issues outweigh any secondary benefits to predators or fisheries. It is recommended the linkages between cormorants [and heron and otter] populations and invasive alien fish species, such as round goby, which have created a new food base, are explored, and the contribution to growth and sustainability of the breeding populations of these predators are evaluated in relation to native prey species and not reliance on non-native species.

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²¹ https://ec.europa.eu/info/funding-tenders/opportunities/portal/screen/opportunities/tender-details/docs/ea46d6ab-c0df-4066-a4f0-84c303637730-CN/0021%20Great%20Cormorants_D3_03.06.26_final_V1.pdf

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9.2 Pygmy cormorant - *Microcarbo pygmaeus*, syn *Phalacrocorax pygmaeus*

9.2.1 Distribution and abundance

The pygmy cormorant has restricted distribution in Europe. There are potentially two breeding populations, one in south-east Europe (east from Italy) around the Black Sea and Mediterranean, and another around the Caspian Sea and south-west Asia (Figure 9). The European population is estimated at 84 600-111 000 mature individuals, with 42 300-55 500 breeding pairs and 53 300-134 000 non-breeding individuals (BirdLife International 2021; <https://datazone.birdlife.org/search>). The species is listed as Least Concern on the IUCN Red List and the overall population trend in Europe is considered to be increasing (BirdLife International 2021).



Figure 9. Distribution of pygmy cormorant in Europe highlighting location of resident ●, breeding ● and non breeding ● areas and discrimination of discrete populations (dotted line) (Source: <https://criticalsites.wetlands.org/en>)

9.2.2 Habitat and ecology

The species occurs in reedbeds, transition zones between reedbeds and open waters, extensively grazed or mowed shores and wet meadows and, in winter, in coastal wetlands, along rivers, and sometimes on inland lakes (Johnsgard 1993, Crivelli et al. 2000, BirdLife International 2004). The species breeds between April and July in large mixed-species colonies, leaving the breeding grounds towards the end of August and returning between March and April (del Hoyo et al. 1992).

The species is sedentary over much of its range with some populations migrating over short distances (Figure 10). Most migration is short-distance movements south from breeding grounds to open waters and wetlands during winter.

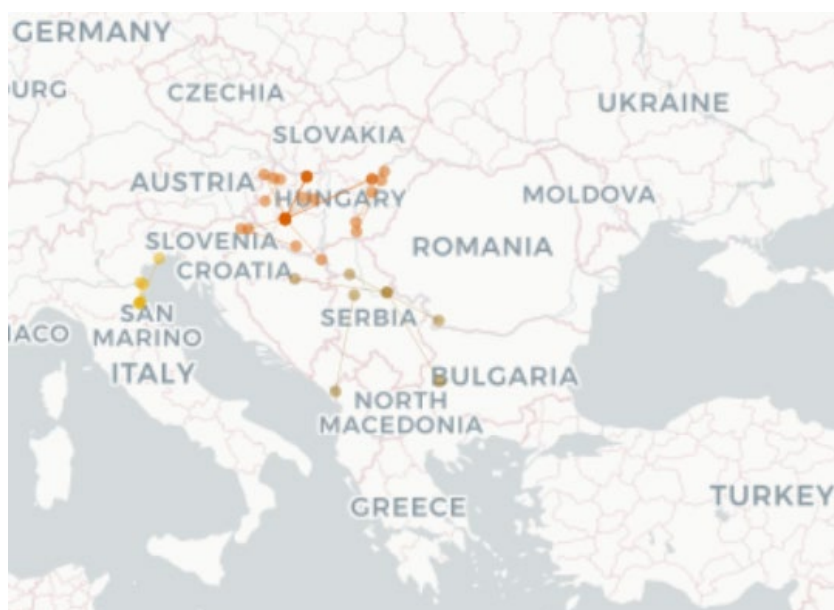


Figure 10. Migration patterns of pygmy cormorant from The Eurasian African Bird Migration Atlas Source: Bird migration atlas: great cormorant - <https://migrationatlas.org/index.php/node/1585>

Throughout the year it normally feeds singly or in small groups (Nelson et al. 2005). Diet consists predominantly of fish up to 15 cm long), including small cyprinids, perch and gobies (del Hoyo et al. 1992). Pygmy cormorants actively dive for prey in managed and wild inland waters, making them highly vulnerable to being caught as bycatch in gillnets, traps, and shallow-water fishing gears.

9.2.3 Ecological and economic impact of predation

Pygmy cormorants are efficient fish predators, but their smaller body size limits their daily food consumption relative to great cormorants. Their impacts are typically localized, affecting lakes, reservoirs, floodplain wetlands and fish ponds rather than large river basins. Evidence that they cause widespread declines in freshwater fish populations at regional scales is limited, largely because of the lack of studies targeting this species. Conflicts with fisheries are most likely where bird numbers concentrate in shallow waters containing dense populations of juvenile fish and on fish farms. While smaller in body size compared to the great cormorant, its foraging habits make it a problem species for commercial fish producers: commercial grow-out ponds, fish nurseries, and fingerling stocks being exceptionally vulnerable.

Pygmy cormorants frequently forage in high numbers around artificial ponds and reservoirs, reducing commercial yields. In these circumstance their impact becomes a major financial challenge for producers and a driver of illegal persecution and bird mortality. This financial challenge is reflected in the limited data collected from the EIFAAC/EEA/FEAP survey (FAO, 2026b) (Table 5).

Only six countries reported damage, but total losses are in excess of EUR 13 million per annum, especially in Cyprus, Greece, Romania and Serbia. This is a limited survey and countries like Poland that have applied for numerous Article 9 derogations have not reported; thus economic damage from pygmy cormorant is likely to be considerable in other EU and non EU countries.

Table 5. Regional overview of average damage by pygmy cormorant on fish populations in European countries (source FAO, 2026b)

Country	Pygmy cormorant population (individuals)	Occurrence in the year (days)	Estimated damages caused (EUR, average annual)	Reference years
Croatia	21	90	4252.5	2023-2024
Cyprus	2250	180	928 500	2023-2024
Greece	750	365	6 160 000	2013-2018
Republic of Moldova	100	360	81 000	2023-2024
Romania	16 000	210	5 348 500	2023-2024
Serbia	1600	365	1 314 000	2023-2024
Total	20 721	262	1 3836 252	2023-2024

9.2.4 Mitigation measures

The pygmy cormorant has received increasing attention over the last decade because its population has expanded across parts of south-eastern and central Europe under legal protection. Compared with the great cormorant (*Phalacrocorax carbo*), however, there are relatively few studies that specifically quantify its ecological or economic impacts on freshwater fisheries or aquaculture. Most publications examine its ecology, distribution, conservation and interactions with wetlands rather than documenting fishery losses.

The species is listed on Appendix I of the Convention on Migratory Species, and is covered by the African Eurasian Waterbird Agreement. It is listed under Annex II of the Bern Convention and Annex I of the EU Birds Directive. Consequently any actions in the EU must align with these agreements and tend to be orientated around coexistence rather than predator control, including restoring structurally diverse fish habitats and using pond netting where local conflicts occur. Nevertheless, actions are taken to manage pygmy cormorant populations, mostly in Poland, largely for protection of other flora and fauna and capture and release for research (Figure 11). Lethal measures were limited and only applied in Poland.

In south-east Europe, conservation measures have ameliorated the most important threats to pygmy cormorants (Crivelli et al. 2000), although concern still exists over persecution in its wintering range. Elsewhere the species is hunted because of its impact on aquaculture units (Eken and Magnin 1999, Kazantzidis and Nazirides 1999, Crivelli et al. 2000). Farmers frequently use lethal control or non-lethal scaring methods to protect their stock. The evidence available indicates that pygmy cormorants can have locally critical ecological effects on juvenile fish communities in shallow freshwater habitats and fish ponds, but there is little evidence that they cause widespread regional declines in freshwater fisheries, although considerable economic losses to fisheries and aquaculture are reported (Table 5). Current European research therefore views the species as a native wetland predator whose effects should be managed through targeted, site-specific mitigation.



Figure 11. Number of derogations issued under Article 9 of the Birds Directive to control pygmy cormorants in Europe: 2023–2025. Source: European Environment Agency (2024a) Overview of derogations and exceptions to species protection across the EU. <https://www.eea.europa.eu/en/analysis/maps-and-charts/overview-of-derogations-and-exceptions-dashboards>. Cited 07 August 2026

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9.3 Grey heron – *Ardea cinerea*

9.3.1 Distribution and abundance

The grey heron is widely distributed in Europe (Figure 12). There are potentially two breeding populations in Europe, one in Western Europe and north-west Africa and second in south and Eastern Europe (east from Italy) around the Black Sea and eastern Mediterranean. There is also another population around the Caspian Sea.

The global population of grey heron is estimated to number ≈790 000-3 700 000 individuals (Wetlands International 2015). The European population is estimated at 223 000-391 000 pairs, which equates to 447 000-782 000 mature individuals (BirdLife International 2015; <https://datazone.birdlife.org/search>). The species is listed as Least Concern on the IUCN Red List and the overall population trend in Europe is considered to be increasing (BirdLife International 2021).

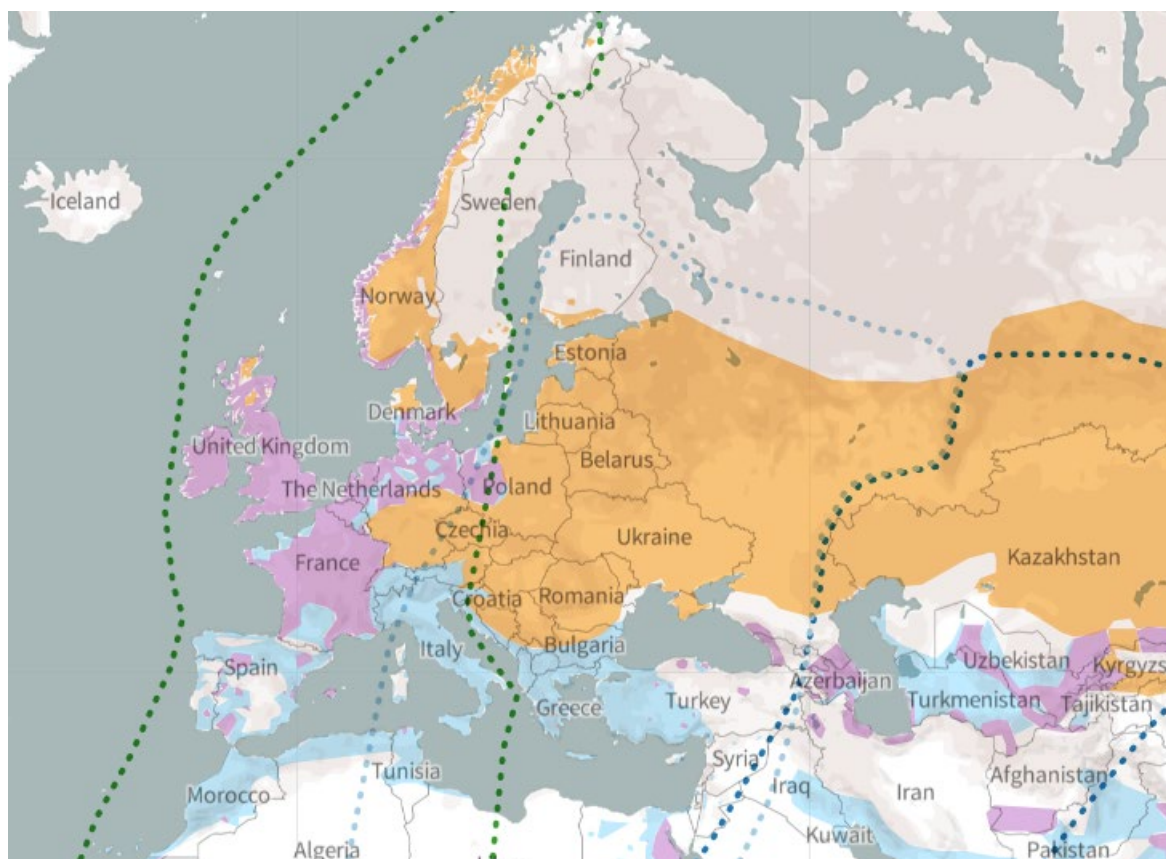


Figure 12. Distribution of grey heron in Europe highlighting location of resident ●, breeding ● and non breeding ● areas and discrimination of discrete populations (dotted line) (Source: <https://criticalsites.wetlands.org/en>)

9.3.2 Habitat and ecology

The grey heron occupies a broad range of shallow water habitats including wide rivers, lake shores, ponds and floodplains habitats (Brown et al. 1982; Snow and Perrins 1998; Kushlan and Hancock 2005). The species has a preference for areas with trees as it is commonly an arboreal rooster and nester.

Its movement patterns vary from local dispersal of hundreds of kilometres to true migrations. Migration is more common in northern and eastern populations (Manikowska-Slepowska et al. 2018). Overall, migration takes place along a NE-SW axis, including into North Africa. This is mainly populations moving south to overwinter (Campos et al. 2001): Northern European grey herons overwinter mostly in the British Isles and central Europe. Birds from the British Isles and west and south-west Europe are generally resident (Campos et al. 2001; Wernham et al. 2002), but with some NE-SW movements as far as Africa.

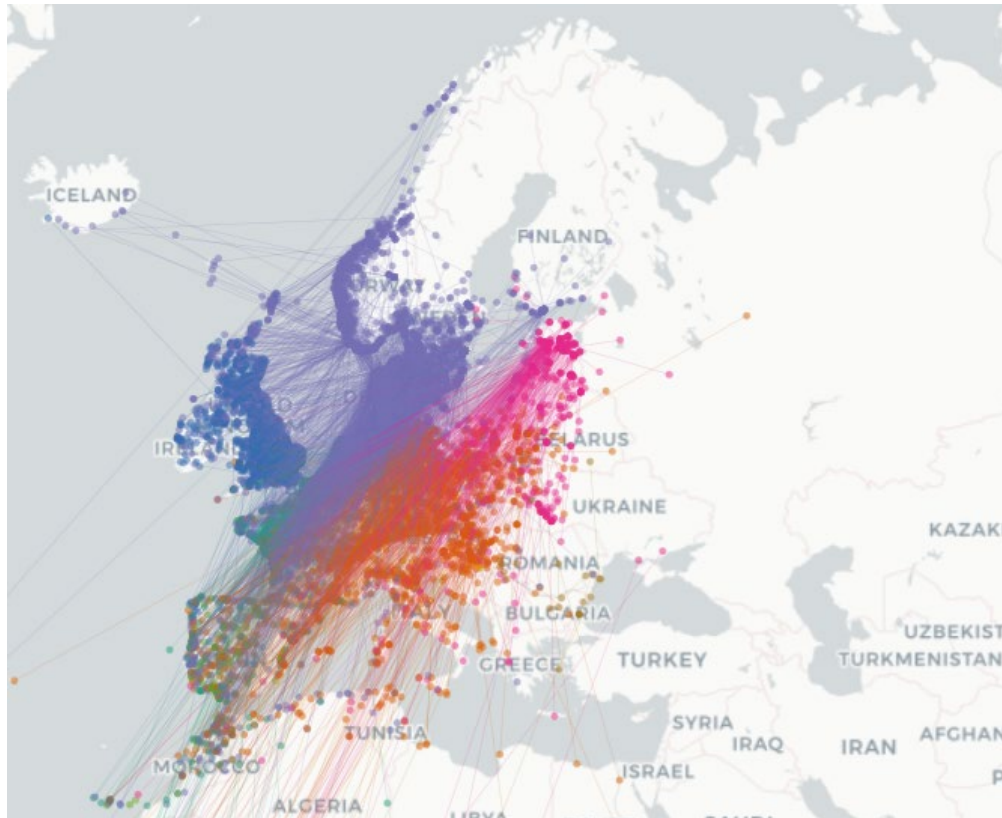


Figure 13. Migration patterns of grey heron from The Eurasian African Bird Migration Atlas Source: Bird migration atlas: great cormorant - <https://migrationatlas.org/index.php/node/1794>

The grey heron is a solitary, visually oriented predator that typically forages in shallow water using a sit-and-wait or slow-stalking strategy. Their foraging success is strongly influenced by water depth, turbidity and prey density. Grey herons are constrained by their morphology and hunting strategy, limiting them to relatively shallow habitats. This spatial restriction shapes their ecological role, concentrating predation pressure in littoral zones and small water bodies, including fish ponds.

Fish dominate the diet of grey herons, often comprising over 80% of prey items in inland systems (Marquiss and Leitch, 1990). Common prey species include small cyprinids, perch, and juvenile salmonids. Herons generally select prey based on size and accessibility, favouring small to medium-sized fish that can be captured and swallowed whole, which may have implications for fish recruitment dynamics. Herons are opportunistic and will consume amphibians, invertebrates and small mammals when fish are less available.

Grey herons exhibit flexible foraging behaviour, often exploiting habitats with high prey density. They are known to concentrate in areas such as fish farms, stocked lakes, and shallow river margins, where prey is abundant and easily accessible (Carss, 1993; Ekblad et al. 2026). Fish farms and managed fisheries often present artificially high prey densities, leading to elevated predation rates that may not be representative of natural systems (Ekblad et al. 2026). Heron foraging behaviour is influenced by

temporal factors such as season and time of day. For example, increased predation may occur during winter, when alternative prey is scarce and fish are more vulnerable due to reduced mobility.

9.3.3 Ecological and economic impact of predation

Predation by piscivorous birds represents a persistent challenge for inland fisheries and aquaculture systems across Europe. Among these predators, grey heron occupies a significant ecological niche as an opportunistic feeder in freshwater environments. While they play a role in maintaining ecosystem balance, their interaction with managed fish stocks has raised concerns among fishery managers and aquaculture operators.

From an ecological perspective, herons exert direct and indirect influences on fish populations. As visual predators, they typically target small to medium-sized fish in shallow waters, often selecting individuals that are slower or more exposed. This selective predation can alter the size and age structure of fish populations, particularly in stocked fisheries where juvenile fish are abundant and vulnerable. In confined or semi-natural systems such as ponds and small lakes, repeated predation events may lead to localized depletion of fish stocks. As a consequence herons can have significant local impacts, particularly on stocked fish (Carss, 1993). In such contexts, predation may result in measurable economic losses. Unlike colonial species, herons are predominantly solitary hunters, which limits the spatial scale of their impact and reduces the likelihood of widespread ecosystem-level depletion. In rivers and lakes with complex habitats and diverse fish communities, heron predation is generally considered to be compensatory rather than additive, targeting vulnerable individuals that may have lower survival probabilities.

At the farm level, impacts can vary widely depending on site characteristics, but documented cases indicate substantial losses in both biomass and revenue (EIFAC, 1988; Ekblad et al., 2026). Shallow ponds, which facilitate easy access for wading birds, are especially vulnerable, as are facilities located near heron nesting colonies.

Herons also play an important regulatory role within freshwater ecosystems. By preying on fish, amphibians, and invertebrates, they contribute to trophic balance and biodiversity maintenance. Their predation may help control overabundant species and remove diseased or weakened individuals, potentially enhancing overall stock health. Consequently, their ecological role cannot be considered solely detrimental; rather, it reflects a natural component of aquatic food webs. Nonetheless, indirect ecological effects can arise from their presence. Fish exposed to predation risk often exhibit behavioural changes, such as reduced feeding activity or increased shelter use, which can lead to slower growth rates and decreased fitness (Carss, 1993). These behavioural responses may interact with other environmental stressors, amplifying their ecological significance.

The economic implications of heron predation are particularly pronounced in aquaculture and recreational fisheries (Ekblad et al., 2026). Direct financial losses occur through the consumption of stocked or cultivated fish, with estimates suggesting that heron predation contributes to tens of millions of euros in annual losses across Europe. This is exemplified by the limited data collected from the EIFAAC/EEA/FEAP survey (FAO 2026b) (Table 6). Thirteen countries reported damage, with losses are in excess of EUR 22 million per annum. This was a limited survey but the massive losses in countries like Poland (in excess of EUR 9 million losses per annum) justify the high number of Article #9 derogations approved. The economic damage from herons is also likely to be considerable in other EU and non EU countries.

Table 6. Regional overview of average damage by grey heron on fish populations in European countries (source FAO, 2026b)

Country	Grey heron population (individuals)	Occurrence in the year (days)	Estimated damages caused (EUR, average annual)	Reference years
Austria	1505	365	366 960	2023-2024
Belgium	312	360	219 000	2023-2024
Croatia	397	365	138 934	2023-2024
Cyprus	1050	365	278500	2023-2024
Czechia	6705	365	1 939 640	2023-2024
Finland	3000	300	604 800	2023-2025
Greece	12 200	210	1 750 000	2023-2024
Republic of Moldova	5000	150	960000	2023-2024
Netherlands	25 800	200	9 040 120	2023-2024
Poland	5415	365	1 304 678	2023-2024
Romania	20 440	251	3 644 000	2023-2024
Serbia	4100	210	945 000	2023-2024
Slovakia	1500	240	919 000	2023-2024
Total	86 972	365	22 110 632	2023-2024

In addition to direct consumption, herons cause non-lethal damage that further exacerbates economic losses. Fish that are attacked but not consumed may suffer injuries that reduce their market value or lead to mortality at a later stage. Stress induced by repeated predation attempts can also impair growth rates, extending production cycles and increasing operational costs. These indirect losses are difficult to quantify but may represent a significant proportion of total economic impact.

Mitigation efforts introduce additional financial burdens for aquaculture operators and fishery managers. Preventive measures such as overhead netting, wire systems, and visual or acoustic deterrents require both initial investment and ongoing maintenance. In some cases, habitat modification, such as increasing pond depth or altering bank profiles, is implemented to reduce accessibility for herons. While these strategies can be effective, their costs may match or exceed the value of the losses they are intended to prevent, particularly for small-scale enterprises.

Recreational fisheries are also affected by heron predation. Stocked fish species, such as trout, are highly susceptible to predation due to their behaviour and visibility in the water column. Reduced fish abundance can lower catch rates, diminishing angler satisfaction and potentially affecting revenue from permits and tourism. Consequently, predation pressure can have broader socio-economic implications beyond direct financial losses.

9.3.4 Mitigation measures

The severity of heron predation impacts varies considerably across Europe and is influenced by several key factors. Farm design is critical, with shallow, open systems being more susceptible than deeper or

enclosed ones. Proximity to heronries increases predation frequency, while fish species and stocking density further shape vulnerability. Intensive management practices, particularly in aquaculture, tend to amplify the effects of predation compared to natural or extensively managed systems (Schwerdtner Máñez et al., 2025).

The grey heron is not listed in Annex I, Annex II or Annex III of the Birds Directive, but Article 1 of the Directive covers all species of naturally occurring wild birds in the EU, and Article 5 establishes a general protection regime for them. Consequently any actions in the EU must align with these agreement. It is also covered by Annex II of the African Eurasian Waterbird Agreement. Considerable action are taken to manage grey heron populations under Article 9 derogations (Figure 14). Lethal measures were widespread across countries mostly for the prevention of serious damage.

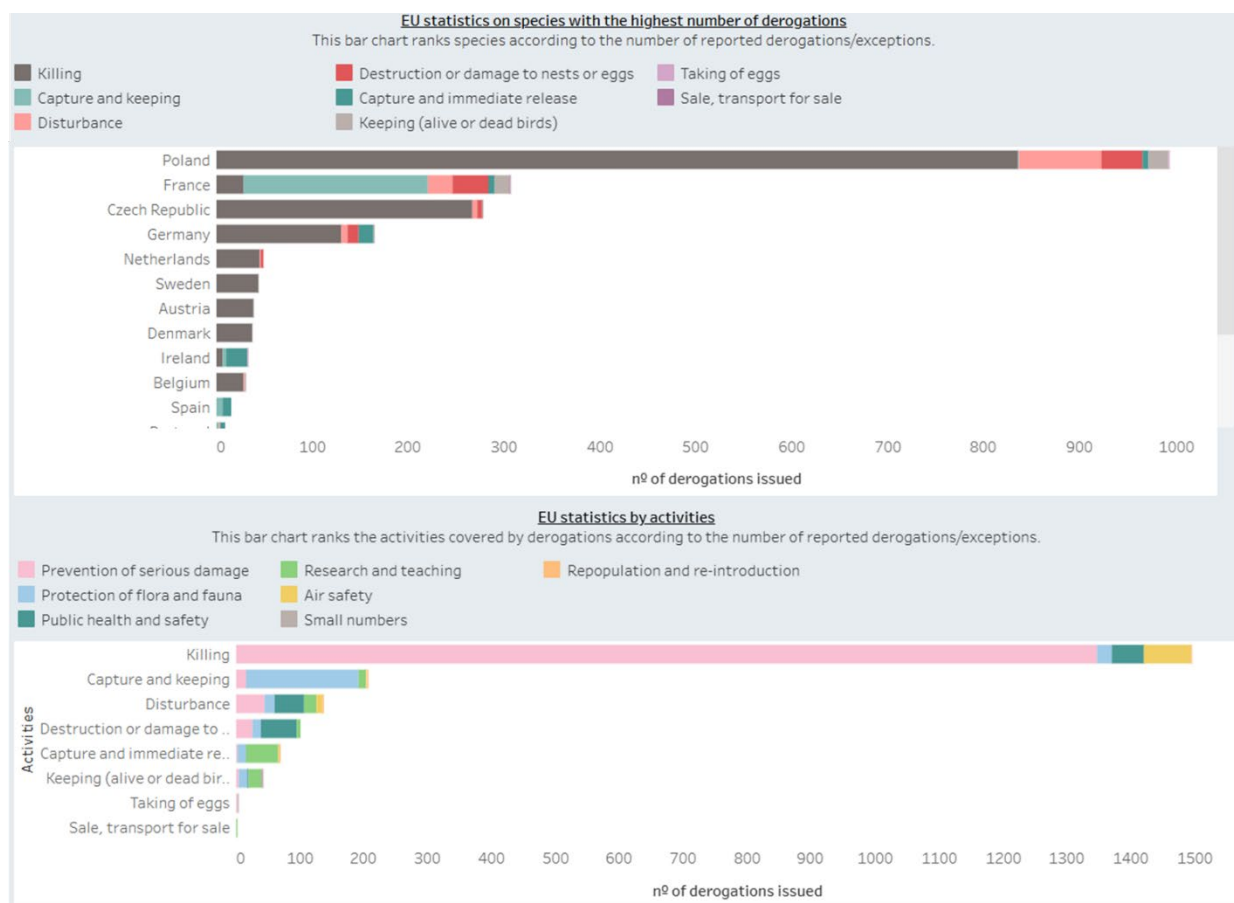


Figure 14. Number of derogations issued under Article 9 of the Birds Directive to control grey herons in Europe: 2020–2025. Source: European Environment Agency (2024a) Overview of derogations and exceptions to species protection across the EU. <https://www.eea.europa.eu/en/analysis/maps-and-charts/overview-of-derogations-and-exceptions-dashboards>. Cited 07 August 2026

Other non-lethal actions are taken against grey herons on fish farm settings where practical, including scaring and exclusion devices such as overhead netting and wires and structural refuges. In more open water systems the most viable option is creating deeper water zones, increasing submerged vegetation cover, constructing refuge structures and creating complex stream and pond margins. This is aimed at reducing accessibility of fish to wading birds.

In conclusion, heron predation on inland fisheries and aquaculture in Europe presents a multifaceted issue with both ecological and economic dimensions. Ecologically, herons function as natural predators that contribute to ecosystem regulation, although they can locally reduce fish populations and influence behaviour. Economically, they impose significant costs through direct consumption,

indirect damage, and the need for mitigation measures. Ultimately, the impact of heron predation is highly context-dependent, varying according to environmental conditions, management practices, and system vulnerability (Schwerdtner Máñez et al., 2025). Addressing this challenge requires site-specific strategies that reconcile conservation priorities with the needs of fisheries and aquaculture activities.

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9.4 Little egret - *Egretta garzetta*

9.4.1 Distribution and abundance

The little egret has restricted distribution in Europe. There are potentially two breeding populations, one in W Europe and NW Africa and a second in central and Eastern Europe, the Black Sea and Eastern Mediterranean (Figure 15). The total population size in Europe is estimated at 121 000-190 000 mature individuals, with 60 800-94 900 breeding pairs and 33 700-47 700 non-breeding individuals (BirdLife International 2021). The population in Europe is considered to have undergone a small decline in recent years, but is categorised as Least Concern on the IUCN Redlist (BirdLife International



2021).

Figure 15. Distribution of little egret in Europe highlighting location of resident ●, breeding ● and non breeding ● areas and discrimination of discrete populations (dotted line) (Source: <https://criticalsites.wetlands.org/en>)

9.4.2 Habitat and ecology

Little egrets inhabit fresh, brackish or saline wetlands and shows a preference for shallow waters (10-15 cm deep) in open, unvegetated sites and where fish are concentrated in pools or at the water surface (Kushlan and Hancock 2005). They are highly migratory (Hancock and Kushlan 1984) and undergo post-breeding dispersive movements (Figure 16). Most movements are linked to short- to medium-distance dispersal patterns often shifting further west and south to overwinter.

The timing of breeding is spring and summer (March to July). The species usually nests in colonies sometimes of thousands of pairs and often with other species (del Hoyo et al. 1992). When not breeding the species commonly feeds solitarily or in loose flocks during the day (del Hoyo et al. 1992).

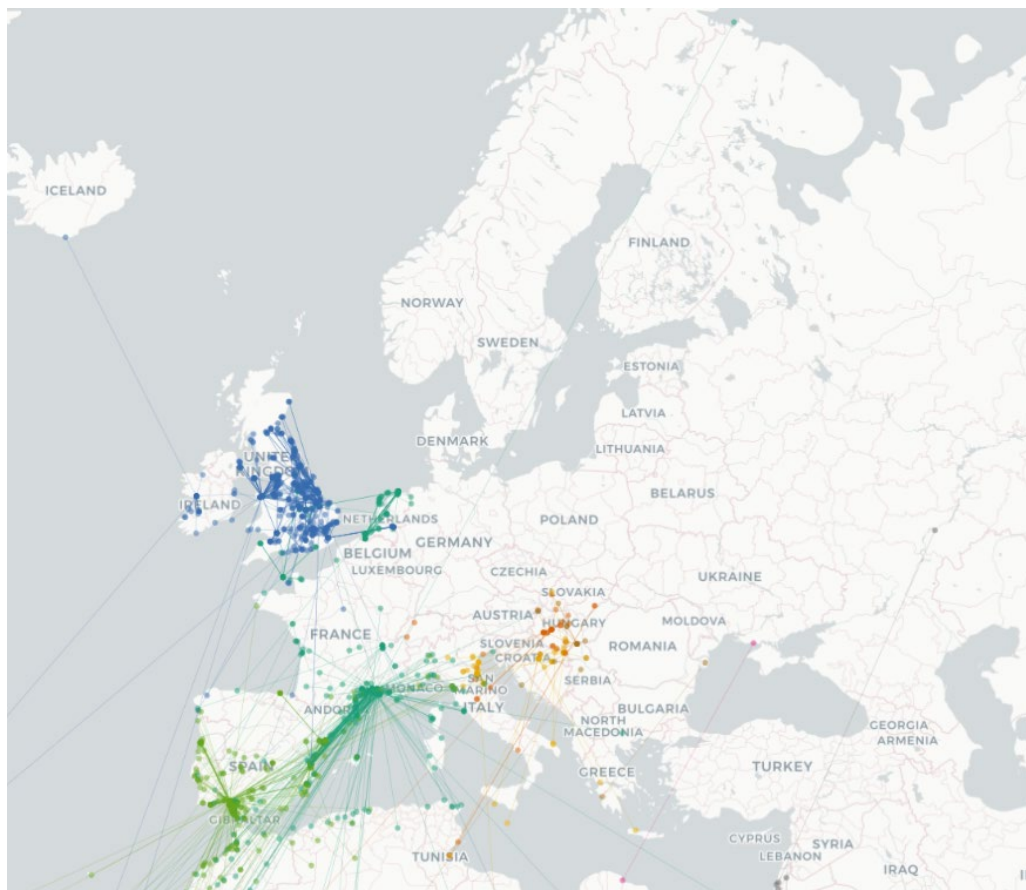


Figure 16. Migration patterns of little egret from The Eurasian African Bird Migration Atlas Source: Bird migration atlas: great cormorant - <https://migrationatlas.org/index.php/node/1665>

Little egrets are visual, shallow-water predators. They typically forage by walking or running through shallow water, disturbing prey with their feet or wings and striking rapidly with the bill. Their diet is dominated by relatively small aquatic prey, particularly small fish under 20 g in weight and less than 10 cm long (averaging 4 cm) (Kushlan and Hancock 2005), amphibians, crustaceans and aquatic insects. Consequently, the species is potentially most important as a predator of young-of-year and juvenile fish, rather than larger fish targeted by recreational or commercial fisheries. There is also evidence individuals preferred commercial fishponds over the other wetland types, presumably because of easier feeding opportunities.

9.4.3 Ecological and economic impact of predation

Little egrets feed on fish in freshwater systems. Their impact is greatest in shallow, clear water with limited fish refugia and with an abundance of juvenile fish. Feeding success is dependent on water turbidity and predation is particularly effective in clear shallow lakes, aquaculture ponds and artificial wetlands (Cezilly, 1992). The greatest threats are likely to occur in extensive carp ponds, especially nursery ponds and shallow grow-out ponds. This is supported from a study in Hong Kong where little egrets have been tracked to show they exploit aquaculture ponds and consume cultured fish (Pang et al., 2020), but there is negligible research on economic losses from egrets in Europe.

Little egrets can also alter prey fish behaviour to avoid predation by egrets but egrets also alter their foraging behaviour in response to prey distribution (Katz et al., 2013).

9.4.4 Mitigation measures

The little egret has received little scientific attention in Europe and no studies specifically quantify its ecological or economic impacts on freshwater fisheries or aquaculture. Most publications examine its ecology, distribution and conservation.

The species is listed on Appendix ii of the Convention on Migratory Species, and is covered by the African Eurasian Waterbird Agreement. It is listed under Annex II of the Bern Convention and Annex I of the EU Birds Directive. It is also listed under Annex A of the EU regulation of trade of fauna and flora. Consequently any actions in the EU must align with these agreements and tend to be orientated around coexistence rather than predator control. Nevertheless actions are taken to manage little egret populations, mostly in France, Poland and Spain, largely for protection of other flora and fauna and capture and release for research (Figure 17). Lethal measures were very limited and only applied in Poland. Other actions include scaring and exclusion devices, such as overhead netting and wires and structural refuges, on fish farm settings where practical. In more open water systems the most viable option is creating deeper water zones, increasing submerged vegetation cover, construct refuge structures and create complex stream and pond margins. This reduces accessibility of fish to wading birds. This may also improve fish welfare and biodiversity.

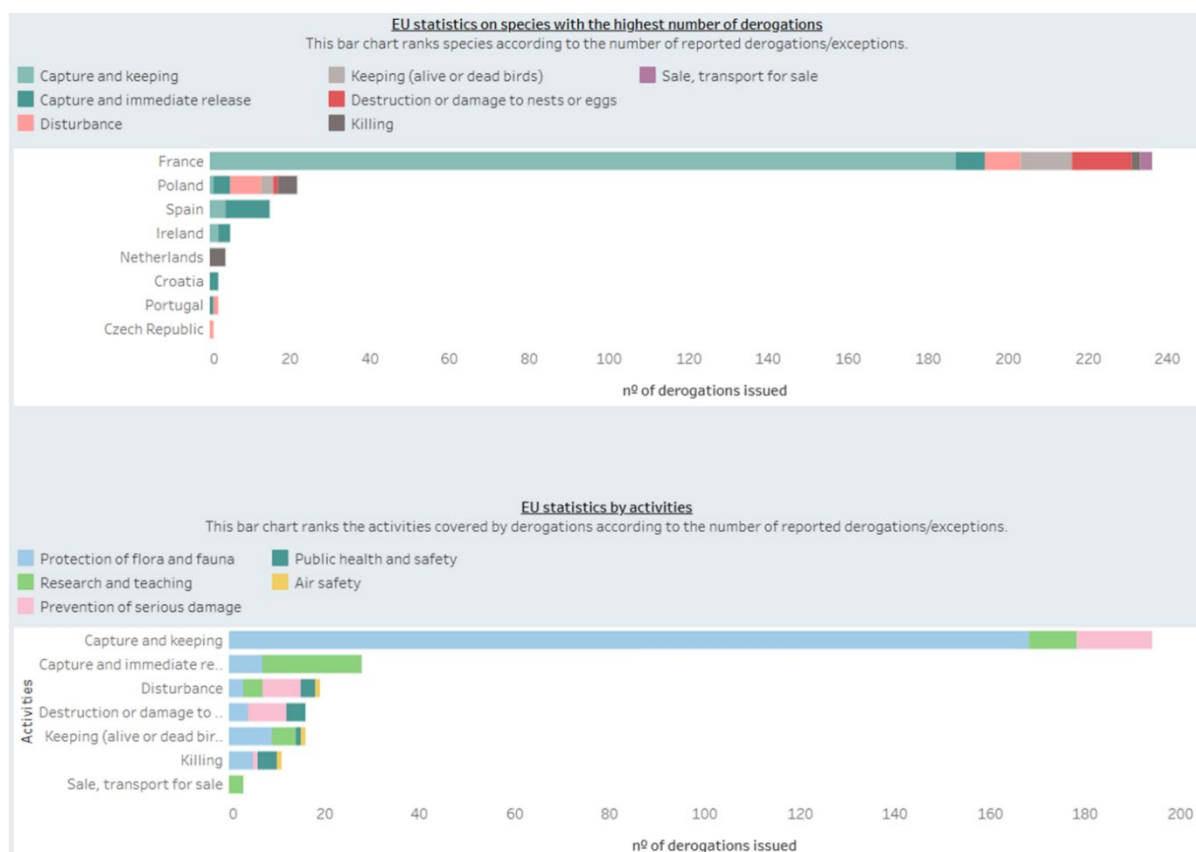


Figure 17 Number of derogations issued under Article 9 of the Birds Directive to control little egret in Europe: 2020–2025. Source: European Environment Agency (2024a) Overview of derogations and exceptions to species protection across the EU. <https://www.eea.europa.eu/en/analysis/maps-and-charts/overview-of-derogations-and-exceptions-dashboards>. Cited 07 August 2026

In conclusion, *Egretta garzetta* is best considered a potentially locally important predator of juvenile and small freshwater fish, particularly in shallow waters and aquaculture ponds, but is not as a major cause of regional freshwater fishery decline.

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9.5 Great white egret - *Ardea alba*; syn *Egretta alba*

9.5.1 Distribution and abundance

The great white egret has restricted distribution in Europe. There is a single European - North African population that inhabits most countries in western, central and southern Europe. (Figure 15). The global population size is estimated at 1 600 000-2 400 000 individuals (Wetlands International, 2023), which equates to approximately 1 100 000-1 600 000 mature individuals. No specific estimate is available for Europe, although the European population is suspected of having undergone an increase in recent years (Lawicki, 2014). The population is listed as Least Concern in Europe on the IUCN Redlist (BirdLife International, 2021).



Figure 18. Distribution of great white egret in Europe highlighting location of resident ●, breeding ● and non breeding ● areas and discrimination of discrete populations (dotted line) (Source: <https://criticalsites.wetlands.org/en>)

9.5.2 Habitat and ecology

The great white egret occupies a broad range of inland and coastal wetland habitats, including river margins, lake shores, marshes, floodplains, aquaculture ponds and reservoirs (del Hoyo et al., 1992; Marchant and Higgins, 1990; Kushlan and Hancock, 2005). The species is highly mobile, with both regular migration and post-breeding dispersal contributing to seasonal movements (Hancock and Kushlan, 1984; Włodarczyk et al., 2020; Figure 19). In Western Europe, birds breeding in France generally migrate northwards, north-eastwards or eastwards after the breeding season. European birds largely remain within Europe during winter, occurring across the Baltic and Atlantic regions from southern Scandinavia to the Iberian Peninsula, with increasing numbers also wintering in the UK.

Great white egrets are colonial breeders, typically nesting in trees near large lakes, reed beds and other extensive wetlands. They generally reach sexual maturity at around 2–3 years of age and form monogamous pairs for each breeding season. Breeding takes place during spring and summer, typically from April to July. Colonies can range from a few tens to several hundred pairs (del Hoyo et

al., 1992; Kushlan and Hancock, 2005). Great white egrets may also nest alongside other waterbird species, forming mixed-species breeding colonies.

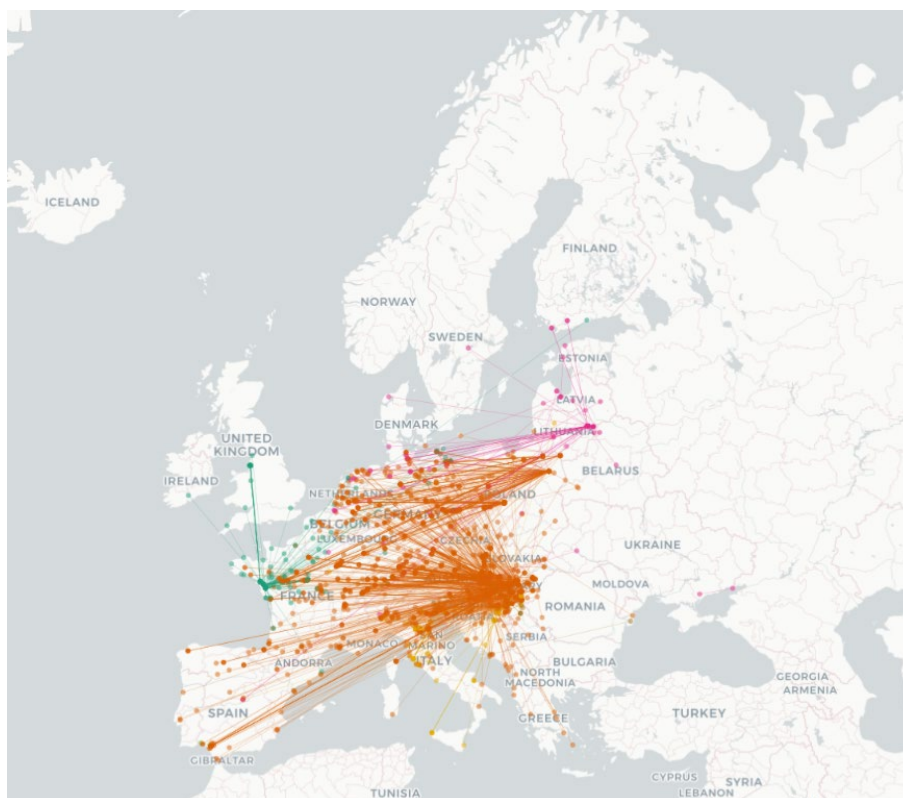


Figure 19. Migration patterns of great white egret from The Eurasian African Bird Migration Atlas
Source: Bird migration atlas: great cormorant - <https://migrationatlas.org/index.php/node/1645>

Outside the breeding season, great white egrets typically forage alone or in small, loosely associated groups, often comprising around 12–50 individuals (Brown et al., 1982; Marchant and Higgins, 1990). Where food is particularly abundant aggregations of several hundred or more birds may form (del Hoyo et al., 1992). The species is primarily diurnal, with foraging activity often greatest around dawn and dusk (del Hoyo et al., 1992; Kushlan and Hancock, 2005). At night, individuals generally roost communally in trees or other suitable vegetation near lakes, rivers and wetlands (Brown et al., 1982).

Great white egrets are efficient predators of shallow-water habitats. Although fish can form an important component of their diet, the species is not exclusively piscivorous and also takes amphibians, crustaceans, aquatic insects, reptiles, small mammals and other aquatic and terrestrial prey (del Hoyo et al., 1992). Compared with the smaller little egret, great white egrets are capable of capturing relatively larger fish, with reported prey lengths of approximately 7.5–10 cm. Their predation is therefore likely to be concentrated primarily on juvenile and small-sized fish, rather than the larger fish generally targeted by recreational and commercial fisheries. There is also evidence that great white egrets preferentially forage at commercial fishponds compared with other wetland habitats, potentially increasing their interactions with managed fish stocks.

9.5.3 Ecological and economic impact of predation

The great egret has received limited scientific attention in Europe and no studies specifically quantify its ecological or economic impacts on freshwater fisheries or aquaculture. Most publications examine its ecology, distribution and conservation.

Great white egrets remove fish from freshwater systems, including aquaculture units, and its predatory behaviour is synonymous with the grey heron (Annex 9.3). Great white egrets are effective predators of small freshwater fish and can exert substantial local predation pressure, particularly in shallow wetlands, fishponds and aquaculture systems. Evidence that this predation produces sustained population-level reductions in wild freshwater fish resources is limited. Economic effects are better documented in the context of extensive pond aquaculture, where great white egrets form part of a broader predator complex that includes cormorants, grey herons and otters. As with little egrets and herons, great white egrets can also alter prey fish behaviour to avoid predation (Katz et al. 2013).

Only one record of predation impact was reported in the EIFAAC/EEA/FEAP (FAO 2026) study where 2300 birds were reported inhabiting Slovakian water bodies for about 180 days in each of 2023 and 2024 causing an estimated EUR 786 000 each year. This is a substantial loss for so few birds.

9.5.4 Mitigation measures

As with little egrets, the great white egret is listed on Appendix ii of the Convention on Migratory Species, and is covered by the African Eurasian Waterbird Agreement. It is listed under Annex II of the Bern Convention and Annex I of the EU Birds Directive. It is also listed under Annex A of the EU regulation of trade of fauna and flora. Consequently any actions in the EU must align with these agreements and tend to be orientated around coexistence rather than predator control. Again synonymous with the little egret actions are taken to manage little egret populations, mostly in France and Poland, but also The Netherlands. Lethal interventions are more prominent management actions for this species than the little egret (Figure 20), which perhaps reflects its great impact on fisheries and aquaculture production units. Other actions include scaring and exclusion devices in fish farm settings where practical.

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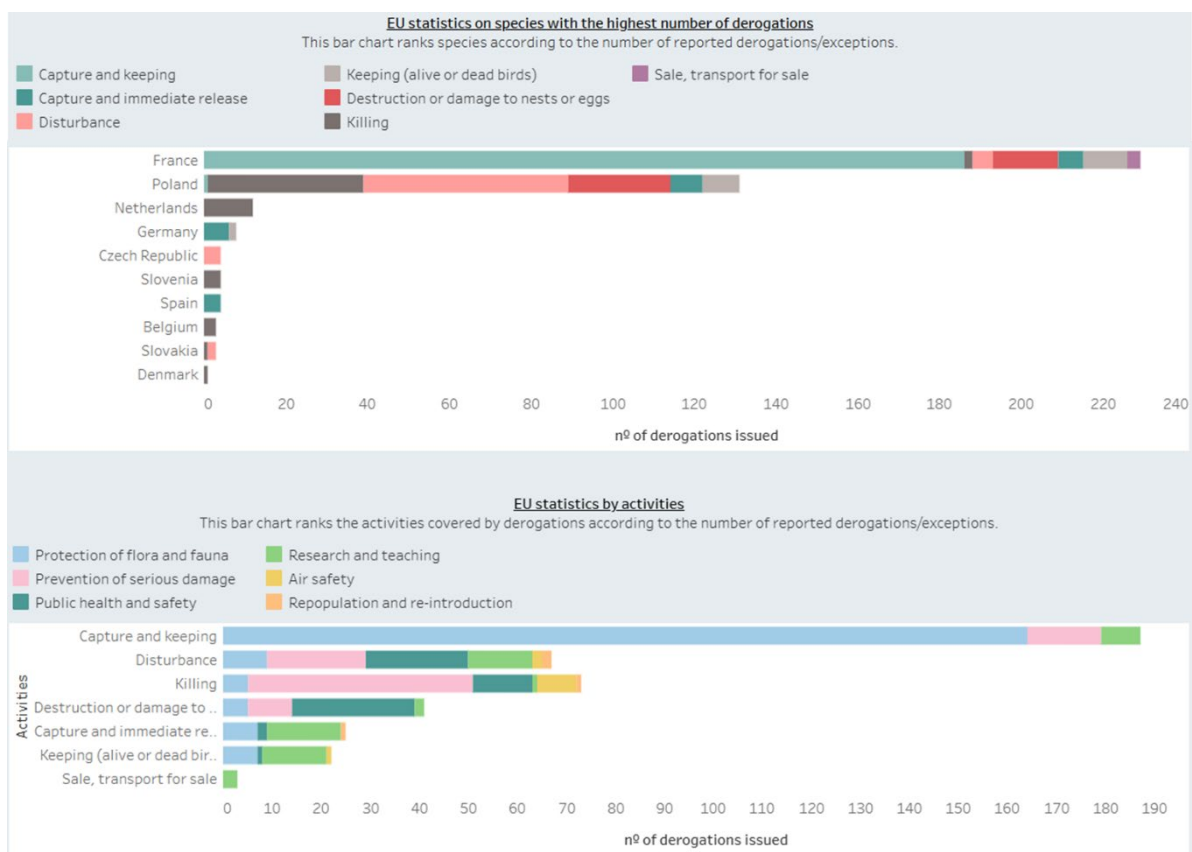


Figure 20. Number of derogations issued under Article 9 of the Birds Directive to control great white egret in Europe: 2020–2025. Source: European Environment Agency (2024a) Overview of derogations and exceptions to species protection across the EU. <https://www.eea.europa.eu/en/analysis/maps-and-charts/overview-of-derogations-and-exceptions-dashboards>. Cited 07 August 2026

9.6 Goosander - *Mergus merganser*

9.6.1 Distribution and abundance

The goosander (*Mergus merganser*) is a highly specialised, piscivorous, diving duck. It is widespread across Europe, inhabiting lakes, reservoirs and slow-flowing rivers. There are potentially a number of populations in Europe, one in north-west and central Europe, one central and west Europe another around the Black Sea, and a separate population in Iceland (Figure 21). The European population of goosander is estimated to be 66 800-103 000 pairs, which equates to 134 000-206 000 mature individuals (BirdLife International 2015). The species is listed as Least Concern on the IUCN Red List and the overall population trend in Europe is considered to be stable (BirdLife International 2015, EBCC 2018).



Figure 21. Distribution of goosander in Europe highlighting location of resident ●, breeding ● and non breeding ● areas and discrimination of discrete populations (dotted line) (Source: <https://criticalsites.wetlands.org/en>)

9.6.2 Habitat and ecology

Goosanders (*Mergus merganser*) are primarily associated with clear, relatively productive freshwater systems, although they also occur in estuaries and sheltered coastal waters, particularly outside the breeding season. Typical habitats include fast-flowing, gravel-bed rivers and large streams, as well as oligotrophic and mesotrophic lakes and reservoirs (Gregory et al., 1997). In northern Europe, breeding generally occurs between March and July, although breeding may take place throughout the year in some regions (Flint et al., 1984; Johnsgard, 1978; Scott and Rose, 1996). Birds typically breed as solitary pairs or in small, loose groups of up to around 10 individuals. Breeding sites are usually associated with large, clear freshwater lakes and pools, as well as the upper reaches of rivers and streams where suitable nesting cavities and foraging habitat are available (Kear, 2005).

Northern European populations are predominantly migratory, whereas populations in more temperate regions are generally sedentary or undertake relatively short-distance movements (Scott and Rose, 1996; Snow and Perrins, 1998; Figure 22). Migration towards wintering areas occurs mainly between October and December (Scott and Rose, 1996). Outside the breeding season, goosanders may aggregate where fish are concentrated, with feeding groups commonly comprising up to approximately 70 individuals (Kear, 2005). Much larger aggregations, numbering several thousand birds, can occur on suitable roosting waters during winter (Snow and Perrins, 1998).

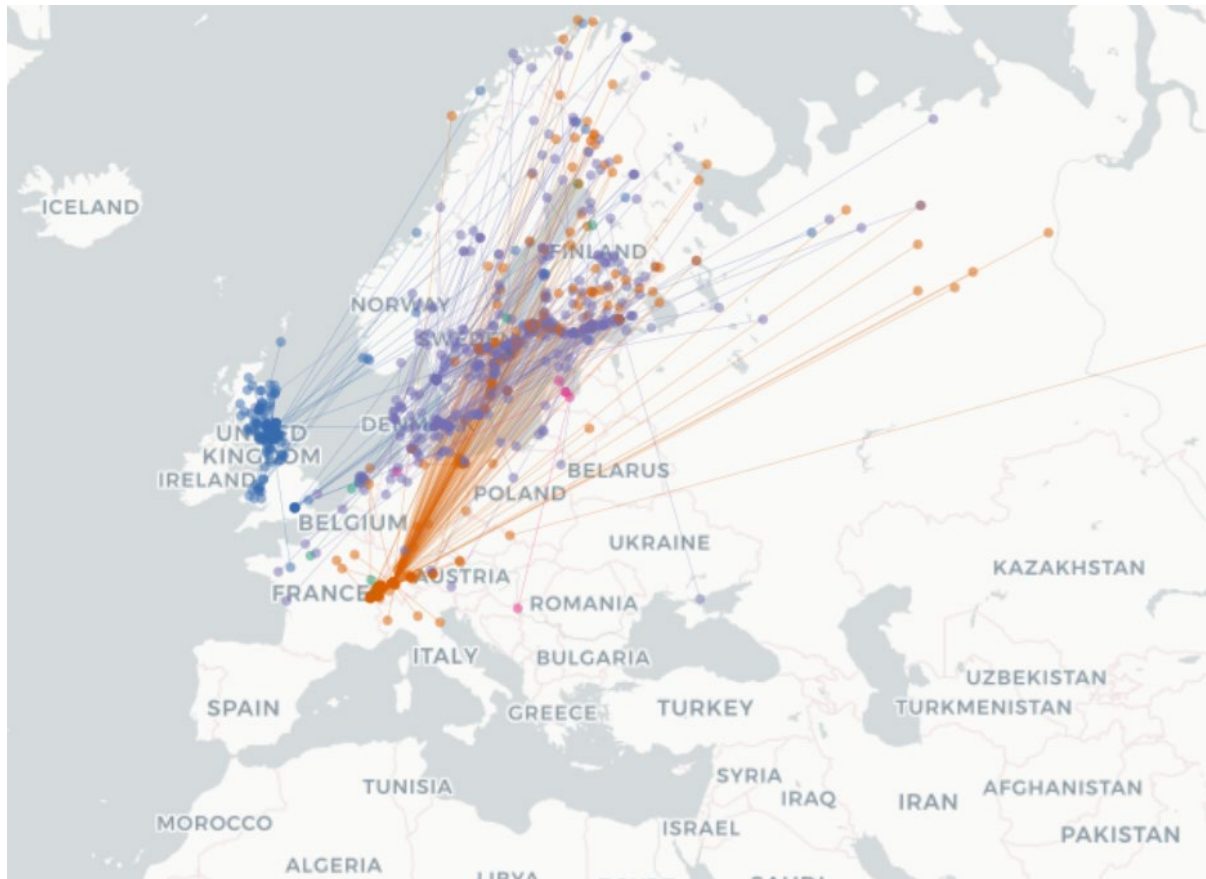


Figure 22. Migration patterns of goosander from The Eurasian African Bird Migration Atlas Source: Bird migration atlas: great cormorant - <https://migrationatlas.org/index.php/node/1602>

Goosanders are highly specialised pursuit-diving predators. Their streamlined body and relatively long neck facilitate efficient underwater movement, while their serrated bills are adapted for grasping and retaining slippery prey. They can pursue fish underwater, including in fast-flowing rivers, and may forage individually or cooperatively when prey is concentrated. Fish form the dominant component of the diet, although goosanders also consume aquatic invertebrates, including molluscs, crustaceans, worms and aquatic insects, together with amphibians, small mammals and birds (Feltham and MacLean, 2005; Kear, 2005).

Goosanders generally select relatively small fish, typically less than 10 cm in length, which can be swallowed whole. Their high dependence on fish means that they can consume substantial quantities of prey, particularly when birds concentrate at sites supporting high fish densities. Using doubly labelled water to estimate energy expenditure, Feltham (1995) estimated that goosanders consumed approximately 480–522 g of fish per bird per day in Scottish rivers. Around two-thirds of this consumption comprised juvenile salmon, equivalent to approximately 10–11 smolts or 48–52 parr per bird per day. These estimates illustrate the potential for substantial predation pressure where large numbers of birds aggregate during periods when vulnerable fish are concentrated.

Salmonids can consequently form an important component of the diet, particularly during the spring smolt migration, and predation may be of particular concern in rivers supporting vulnerable or declining salmon populations. Spring surveys on Scottish rivers have reported salmon parr and smolts contributing approximately 15–50% of goosander diet by mass (Carss, 2022). Predation during the smolt migration has therefore been identified as a potentially important source of mortality for juvenile salmon (Chavarie et al., 2022). However, the dietary composition of goosanders varies among rivers and seasons, and predation should not necessarily be interpreted as being directed specifically towards salmon. For example, Carss (2022) reported that trout constituted approximately 20–40% of the diet, depending on river and season, while minnows accounted for approximately 15–38%. Trout and minnows were particularly important during spring, coinciding with the salmon smolt migration. Consequently, although goosander predation can contribute to mortality of salmon and other valued fish species, the magnitude of its population-level effect will depend on prey availability, bird abundance and aggregation, fish population status, and the extent to which predation is compensatory or additive to other sources of mortality. This distinction is important when assessing potential conflicts between goosander conservation and fisheries management.

9.6.3 Ecological and economic impact of predation

Goosanders are capable of substantial predation on juvenile salmonids and other freshwater fish, and can create local conflicts with fisheries and fish farms (Carss, 2022). They can cause considerable damage feeding on juvenile salmon, especially the parr (e.g. Feltham et al., 1999) and smolt stages (Carss, 2022; Chavarie et al., 2022), but the impact of mortality at these juvenile life stages on declining salmon populations is difficult to determine, given the considerable other pressures acting on the populations.

The relevance of goosander predation to fish-population sustainability is likely to vary spatially and temporally according to predator abundance, prey availability, river characteristics and the size and productivity of the fish populations (Carss, 2022; Chavarie et al., 2022; Feltham et al., 1999; Suter, 1997; Wilson et al., 2007). Consequently, the presence of substantial predation does not necessarily translate directly into a population-level effect, and the ecological and fisheries significance needs to be assessed in the context of other sources of mortality and the capacity of affected fish populations to compensate through growth and recruitment. Notwithstanding, the high concentration of predation on salmon smolts during the relatively short, spring downstream migration could potentially have considerable additional mortality that could impact on salmon population sustainability (Chavarie et al., 2022; Kålås et al., 1993; Wilson et al., 2007). This is likely the primary reason for the considerable concern over goosanders during the March to May smolt migration period and where management efforts need to focus.

There is little evidence for direct economic impacts on fisheries and commercial aquaculture, although the EIFAAC/EEA/FEAP survey (FAO 2026c) reported 750 mergansers foraging for 200 days in 2024 and causing an estimated EUR 337 500 of damage. Goosanders can also take cultured fish, particularly small trout and other fish, within their preferred prey size, but there is no definitive evidence of impact.

As with other fish-eating predators, indirect impacts are evident as a result so fish being injured or stressed avoiding being eaten. This may also alter behaviour in response to predator presence and reduce feeding activities to avoid predators (Dionne and Dodson, 2002).

9.6.4 Mitigation measures

As with all birds, goosanders are protected under the EU Birds Directive. Consequently any actions in the EU must align with these agreements and tend to be orientated around coexistence rather than predator control. Actions are taken to manage goosander populations, mostly in France, Poland and Finland. Most actions are to protect flora and fauna and prevention of serious damage. Disturbance is a common measure but lethal interventions are prominent in Finland (Figure 23) and perhaps reflect its great impact on fisheries and aquaculture production units. Licences to shoot sawbills, including goosanders, as an aid to scaring have been issued on many Scottish river catchments since 2017 because of concerns about damage to salmon fisheries (Calladine et al., 2026), but management in Scotland suffers from lack of robust estimates of wintering goosander abundance at the catchment scale.

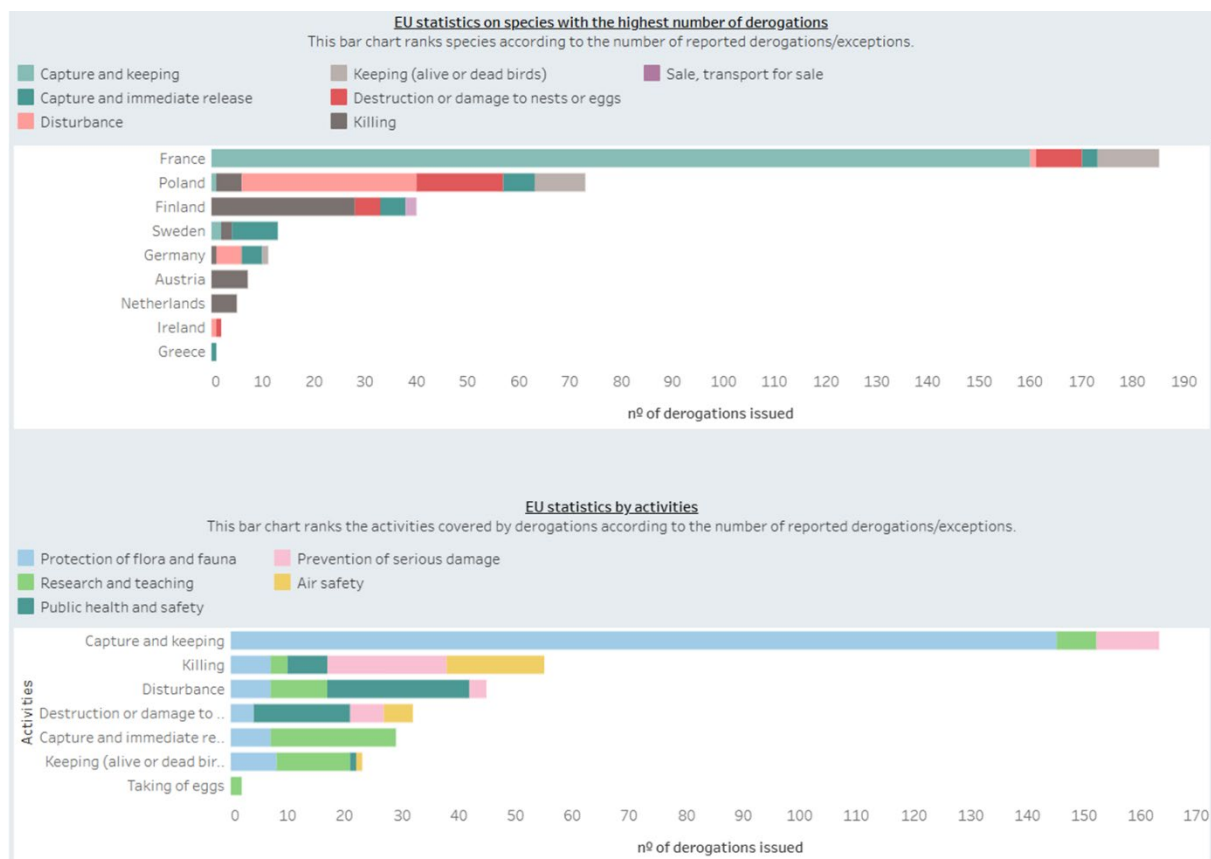


Figure 23. Number of derogations issued under Article 9 of the Birds Directive to control great white egret in Europe: 2020–2025. Source: European Environment Agency (2024a) Overview of derogations and exceptions to species protection across the EU. <https://www.eea.europa.eu/en/analysis/maps-and-charts/overview-of-derogations-and-exceptions-dashboards>. Cited 07 August 2026

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9.7 Red-breasted merganser - *Mergus serrator*

9.7.1 Distribution and abundance

The red-breasted merganser (*Mergus serrator*) is a highly specialised piscivorous diving duck. It is widespread across Europe, inhabiting lakes, reservoirs, and slow-flowing rivers. There are two populations in Europe, one in north-west and central Europe and one in the Black Sea and eastern Mediterranean (Figure 24). The European population of red-breasted merganser is stable at 70 100-120 000 pairs, which equates to 140 000-240 000 mature individuals (BirdLife International, 2015). The species is listed as Least Concern on the IUCN Red List and the overall population trend in Europe is considered stable (BirdLife International, 2015, EBCC, 2018).

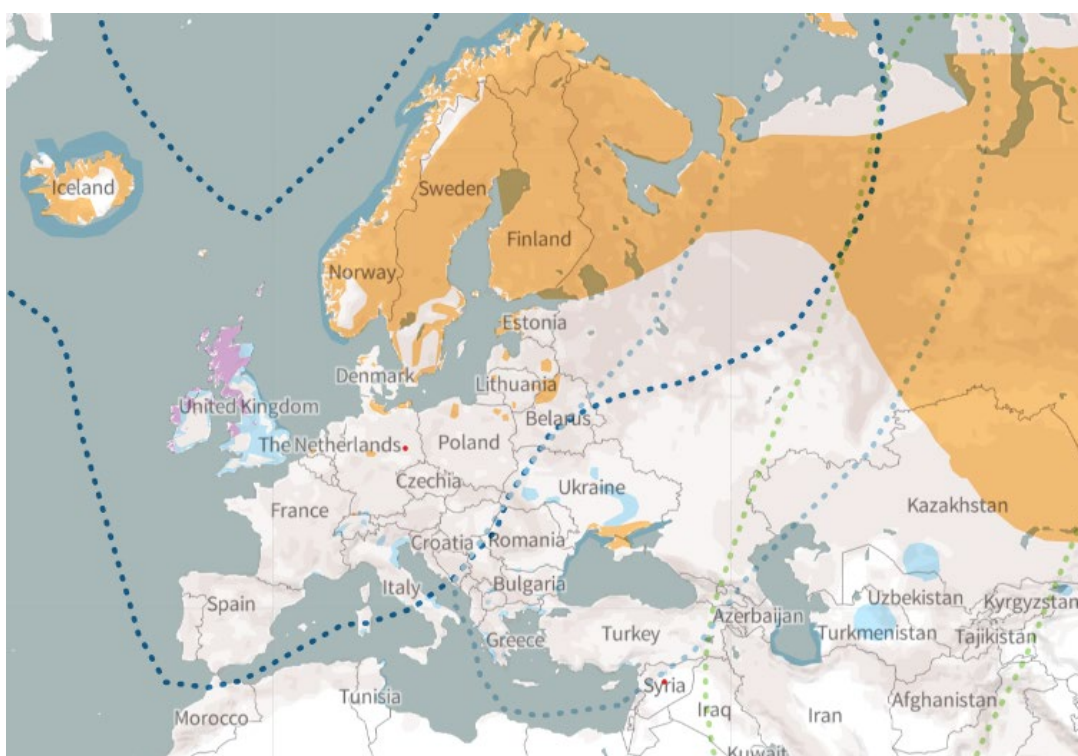


Figure 24. Distribution of red-breasted merganser in Europe highlighting location of resident ●, breeding ● and non breeding ● areas and discrimination of discrete populations (dotted line) (Source: <https://criticalsites.wetlands.org/en>)

9.7.2 Habitat and ecology

Mergansers are distinctive among avian fish predators because they are specialised pursuit divers, using underwater swimming to locate, pursue and capture fish. This differs from wading predators such as the grey heron (*Ardea cinerea*), which primarily use visual ambush techniques from shallow water or along the margins of rivers, lakes and other waterbodies. Mergansers possess a suite of morphological adaptations for underwater predation, including streamlined bodies, relatively long necks and serrated bills that enable them to grasp and retain slippery fish. They are consequently particularly associated with clear rivers and lakes, where good underwater visibility facilitates their visually guided pursuit of prey (Gregory et al., 1997).

Red-breasted mergansers (*Mergus serrator*) breed in a wide range of freshwater and coastal habitats. Inland breeding sites include the wooded shores of deep lakes, as well as smaller rivers and streams (del Hoyo et al., 1992; Kear, 2005). In coastal areas, they may breed around sheltered marine inlets,

estuaries and sandy coastal habitats (Snow and Perrins, 1998). The species is migratory across much of its range (Figure 25), although populations in more temperate regions may undertake relatively short-distance movements to nearby coastal waters or remain close to their breeding areas throughout the year (Scott and Rose, 1996; del Hoyo et al., 1992).

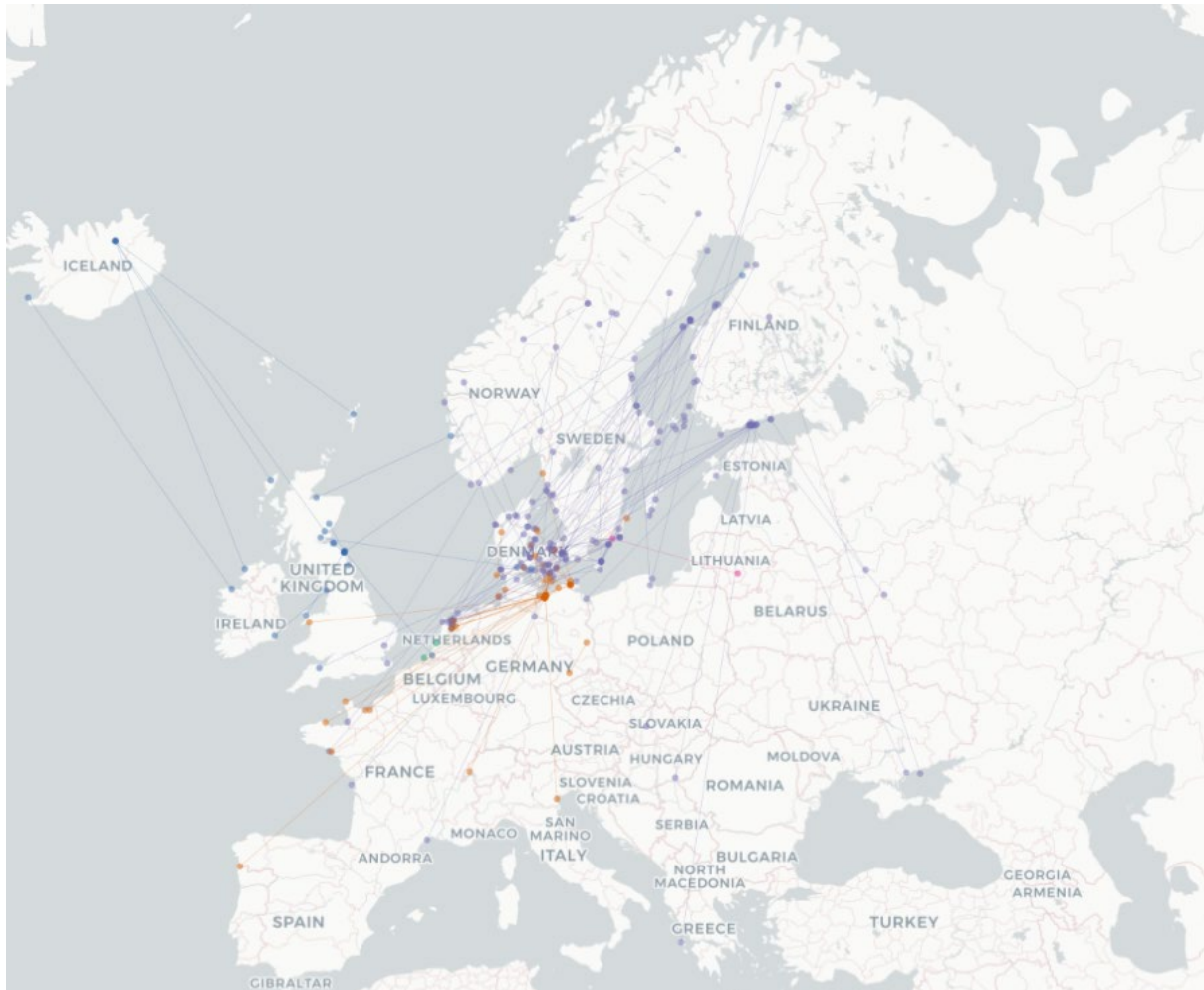


Figure 25. Migration patterns of red breasted merganser from The Eurasian African Bird Migration Atlas
Source: Bird migration atlas: great cormorant - <https://migrationatlas.org/index.php/node/1623>

Breeding generally begins in April or May, although the timing is later at higher latitudes. Birds typically breed as solitary pairs or in small colonies (del Hoyo et al., 1992; Kear, 2005). Autumn migration generally begins from September, while return movements to breeding areas can commence as early as February (Scott and Rose, 1996). During migration and winter, red-breasted mergansers become more gregarious and may form feeding flocks of more than 100 individuals where suitable concentrations of prey occur. In contrast, spring migration is generally undertaken in smaller groups (Snow and Perrins, 1998). This seasonal change in social behaviour reflects the species' tendency to aggregate where food resources are concentrated, potentially increasing localised predation pressure on fish populations during migration and winter.

Fish dominate the diet of red-breasted mergansers and may constitute almost the entire diet in some freshwater systems (Craig et al., 2015; Feltham, 1995; Feltham et al., 1999; Harris et al., 2008). They generally select small- to medium-sized fish, typically 5–20 cm in length. Aquatic invertebrates, including shrimps, crayfish, worms and aquatic insects, may also be consumed, although they normally constitute a relatively small proportion of the diet (Johnsgard 1978; Kear 2005). Their pursuit-diving

behaviour allows them to exploit schools or concentrations of small fish, including juvenile salmonids and small-bodied coarse fish (Sjöberg, 1988).

Of particular fisheries concern is predation on juvenile migratory salmonids. In rivers supporting salmon populations, downstream migration of salmon smolts can temporarily create concentrated and highly vulnerable prey resources and contribute up to 90% of the diet during smolt runs (Feltham, 1995). Red-breasted mergansers may exploit these concentrations, potentially increasing mortality during relatively short periods of the year (Feltham 1995; Feltham and MacLean, 2005). Evidence from Scottish rivers indicates that merganser predation can contribute to mortality of juvenile salmonids during the smolt migration period (Feltham 1995; Harris et al., 2008), and similar concerns have been raised in Welsh rivers. Such predation may be particularly critical where salmon populations are already depleted or exposed to multiple environmental pressures. However, the extent to which predation by red-breasted mergansers translates into reduced recruitment or population-level effects remains uncertain. Population responses depend not only on the number of fish consumed but also on the abundance and productivity of the prey population, compensatory mortality, survival during subsequent life stages and the contribution of other sources of mortality, including habitat degradation, water quality, fishing, disease and changing environmental conditions.

Red-breasted mergansers (and goosanders) also exhibit pronounced spatial and temporal variation in foraging activity. Numbers at freshwater feeding sites may increase substantially during winter as birds move from breeding areas and aggregate at rivers and lakes supporting sufficiently high fish densities. Predation pressure is strongly seasonal and may increase during periods when juvenile fish or migratory fish are particularly abundant and vulnerable. Aggregation of mergansers can further increase local predation pressure because many individuals may exploit the same concentrations of prey simultaneously. This contrasts with predominantly solitary predators such as grey herons, for which predation pressure is generally more spatially dispersed.

The potential fisheries significance of merganser predation therefore depends strongly on where and when predation occurs, rather than simply on the total number of birds present. High densities of birds at productive feeding sites, particularly during periods of concentrated juvenile salmonid migration, may produce substantial local mortality even when predation represents only a relatively small proportion of the total annual mortality of the fish population. Consequently, assessments of the effects of mergansers on fisheries should consider predator abundance, diet composition, prey density, duration of predator–prey overlap and the timing of predation relative to critical fish life stages.

9.7.3 Ecological and economic impact of predation

Red-breasted mergansers can exert substantial predation pressure on juvenile salmonids and other freshwater fish and may therefore generate localised conflicts with fisheries and fish farms (Harris et al., 2008). Particular concern has focused on predation of salmon smolts during their downstream migration, when large numbers of vulnerable fish may pass through areas used by feeding mergansers. High levels of predation have been reported on the River Wye (Feltham et al., 1999) and in Scottish rivers (Feltham, 1995), raising concerns about the potential consequences for salmon recruitment and fisheries production.

In the Scottish study, salmon accounted for approximately 76–91% of the diet by weight during the early stages of the smolt migration (Feltham, 1995). The contribution of salmon subsequently declined as coarse fish became increasingly important in the diet. These observations demonstrate that mergansers can consume large numbers of salmon during periods when smolts are concentrated and

particularly vulnerable to predation. However, high rates of predation during the smolt migration do not, in themselves, demonstrate a population-level effect on adult salmon stocks. Establishing such an effect requires evidence that mortality attributable to predation is additive to other sources of mortality and is sufficient to influence subsequent recruitment and adult abundance.

At present, there is insufficient evidence to conclude that red-breasted merganser predation causes widespread population-level declines in freshwater fish or measurable economic losses to fisheries or aquaculture. Effects are more likely to be localised and seasonal, reflecting variation in bird abundance, fish availability, habitat characteristics and the timing and location of predator–prey interactions. Consequently, substantial predation within an individual river or at a particular time of year should not automatically be interpreted as equivalent to lost fish production at the population or fishery scale.

This distinction is supported by recent assessments of sawbill predation in Scottish rivers, which found no unequivocal evidence linking predation by mergansers and goosanders to population-level effects on fish stocks (Carss, 2022; Calladine et al., 2026). Importantly, the absence of unequivocal evidence should not be interpreted as evidence that no effect occurs. Rather, demonstrating population-level impacts is difficult because long-term data on fish abundance, recruitment and survival are often limited, while information on predator abundance, distribution, feeding locations and prey consumption may also be insufficient. These limitations make it difficult to quantify the proportion of fish mortality attributable to mergansers and to determine whether this mortality is compensatory or additive.

Predation by mergansers is nevertheless recognised as a potential management issue at fish farms. In Scotland, NatureScot guidance²² on licensing arrangements for preventing damage to fisheries and fish farms identifies cormorants, goosanders, mergansers, herons and eider ducks among the species for which licensing applications may be considered where predation occurs or is likely to occur at fish farms. This recognition indicates that interactions between mergansers and farmed fish are considered sufficiently important to warrant management consideration.

A key consideration is therefore the scale at which impacts are assessed. Mergansers may exert significant local predation pressure, particularly in small rivers, at fish-concentration sites, or during periods when birds aggregate and vulnerable juvenile fish are abundant. This does not necessarily translate into a measurable effect on the wider fish population or fishery. Assessments of conflict should therefore distinguish between evidence of predator presence, rates of prey consumption, local mortality, population-level effects and demonstrable economic losses. Making this distinction is essential for proportionate management and for avoiding the assumption that high local predation necessarily results in equivalent reductions in fish production.

²² <https://www.nature.scot/doc/guidance-licensing-arrangements-prevent-damage-fisheries-fish-farms>

9.7.4 Mitigation measures

As with all megansers, red-breasted mergansers are protected under the EU Birds Directive. Consequently any actions in the EU must align with these agreements and tend to be orientated around coexistence rather than predator control. Actions are taken to manage goosander populations, mostly in France, Finland, Poland and the Netherlands. Most actions are to protect flora and fauna and prevention of serious damage. Disturbance is a common measure but lethal interventions are prominent in the Netherlands and Finland (Figure 24) and perhaps reflect its great impact on fisheries and aquaculture production units. Licences to shoot sawbills, including red-breasted mergansers, as an aid to scaring have been issued on many Scottish river catchments since 2017 because of concerns about damage to salmon fisheries (Calladine et al., 2026), but management in Scotland suffers from lack of robust estimates of red-breasted merganser abundance at the catchment scale.

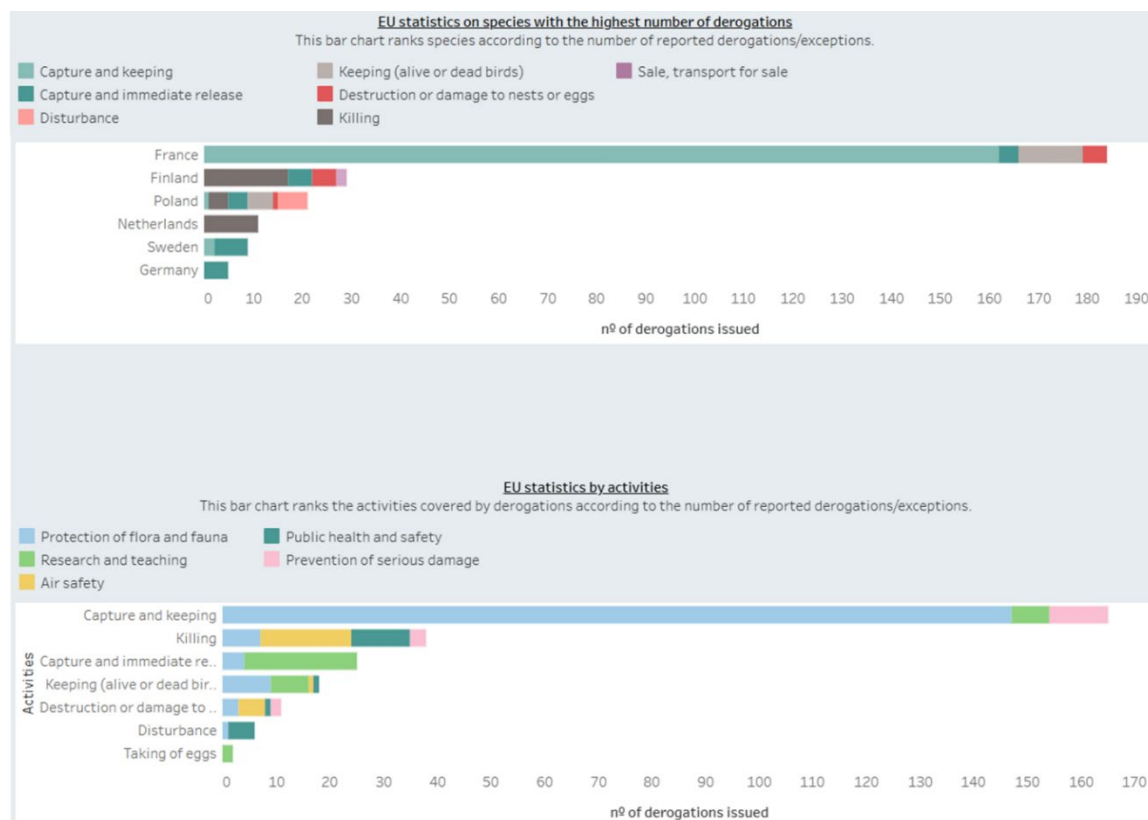


Figure 26. Number of derogations issued under Article 9 of the Birds Directive to control Red-breasted merganser in Europe: 2020–2025. Source: European Environment Agency (2024a) Overview of derogations and exceptions to species protection across the EU. <https://www.eea.europa.eu/en/analysis/maps-and-charts/overview-of-derogations-and-exceptions-dashboards>. Cited 07 August 2026

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9.8 Great crested grebe - *Podiceps cristatus*

9.8.1 Distribution and abundance

The great crested grebe (*Podiceps cristatus*) is widespread across Europe, inhabiting lakes, reservoirs, and slow-flowing rivers. There are potentially two breeding populations in Europe, one in north and Western Europe and the other in the Black Sea and Eastern Mediterranean (Figure 27). The European population is estimated at 330 000-498 000 pairs, which equates to 660 000-997 000 mature individuals (BirdLife International, 2015). The species is listed as Least Concern on the IUCN Red List and the overall population trend in Europe is considered to be declining moderately (BirdLife International, 2015; EBCC, 2018).



Figure 27. Distribution of great crested grebe in Europe highlighting location of resident ●, breeding ● and non breeding ● areas and discrimination of discrete populations (dotted line) (Source: <https://criticalsites.wetlands.org/en>)

9.8.2 Habitat and ecology

The great crested grebe has an average generation length of approximately 7.1 years. It breeds on freshwater and brackish waterbodies with abundant emergent and submerged vegetation. Suitable habitats include small pools and lakes, backwaters of slow-flowing rivers, and artificial waterbodies (del Hoyo et al., 1992; Limona et al., 2018). European populations are generally migratory, although the extent of movement varies considerably among populations, with some undertaking mainly local dispersive movements (Figure 28). Birds breeding in northern and Eastern Europe generally migrate south or south-east towards the Black Sea region, with some populations undertaking relatively long-distance movements. Many birds breeding in Western Europe and Scandinavia migrate south-westwards, with some overwintering in the southern North Sea.



Figure 28. Migration patterns of great crested grebes from The Eurasian African Bird Migration Atlas Source: Bird migration atlas: great cormorant - <https://migrationatlas.org/index.php/node/1706>

The great crested grebe breeds in Europe between April and September. Following breeding, adults may disperse locally and can form large aggregations, sometimes numbering hundreds of individuals (del Hoyo et al., 1992; Fjeldså, 2004). During winter, individuals are generally more dispersed, particularly while feeding, although temporary concentrations of up to several thousand birds may develop at suitable feeding and roosting sites (Snow and Perrins, 1998; Fjeldså, 2004).

Great crested grebes are visually oriented, diving predators, and their foraging efficiency is strongly influenced by water clarity. Successful feeding requires a minimum visibility of approximately 40 cm for effective foraging. Great crested grebe exhibit flock feeding behaviour, especially around breeding areas (Källander, 2008, 2011). Feeding activity may show a crepuscular pattern, with increased activity around dawn and dusk (Piersma et al., 1988). This may coincide with diel movements of fish towards shallower water during periods of reduced light, potentially increasing prey accessibility.

Compared with great cormorants, great crested grebes are constrained by their smaller body size, lower daily food requirements, relatively shallow diving capability and smaller gape. Fish nevertheless form the principal component of their diet, supplemented by aquatic insects, crayfish and molluscs and, occasionally, adult and larval amphibians (Gwiazda, 1997; Gwiazda et al., 2021; Piersma et al., 1988, Ulenaers et al., 1992). Fish consumed are generally relatively small, commonly around 5–15 cm in length, although larger individuals approaching 20 cm may be captured under favourable conditions (Geiger, 1984). Consequently, predation is likely to be concentrated primarily on juvenile and young-of-the-year fish rather than larger adult fish.

The species consumed by great crested grebes vary geographically and according to local fish communities. The predominance of small-bodied and juvenile fish means that great crested grebes are generally more likely to influence recruitment and early life stages than directly reduce the biomass of larger fish available to fisheries. Nevertheless, where large numbers of grebes aggregate at sites with high densities of vulnerable juvenile fish, particularly during concentrated migration periods such as salmon smolt runs, predation could represent a locally important source of mortality. The extent to which this translates into a population-level effect will depend on the magnitude and duration of predation relative to other sources of mortality, prey abundance, recruitment and compensatory processes within the fish population.

The potential significance of predation may be greater in rivers supporting migratory salmonids. During spring, salmon smolts can form relatively concentrated, temporarily abundant prey resources as they migrate downstream. Great crested grebes may exploit these concentrations even where overall fish biomass in the river is relatively low. Consequently, the impact of grebe predation cannot necessarily be inferred from standing fish biomass alone, particularly in salmon rivers where the availability of vulnerable juvenile fish may increase sharply over short periods.

9.8.3 Ecological and economic impact of predation

Predation impact is influenced by a combination of environmental, biological, and anthropogenic factors. Nevertheless, grebes can consume substantial amounts of fish at a local scale. For example, in Lake Como, wintering grebes were estimated to consume between 640 and 1000 kg of fish per winter, equivalent to 10–20% of the commercial fish catch (Martinoli et al., 2003). Because grebes primarily target juvenile fish, their predation may influence recruitment although fish populations can potentially compensate through density-dependent processes such as increased growth and survival among remaining individuals. Consequently, they exert much lower predation pressure on fish biomass. In large rivers and lakes, predation generally represents only a small fraction of annual fish mortality.

Although grebes consume measurable quantities of fish, their economic impact on fisheries is generally considered low. In most cases, consumption represents only a small proportion of total fish production (Martinoli et al., 2003), and conflict with recreational or commercial fisheries is relatively uncommon. Unlike cormorants or herons, great crested grebes rarely cause substantial damage at fish farms. Potential impacts occur mainly at ornamental ponds and extensive carp ponds (Ulenaers and van Vessem, 1994). Where grebes regularly visit ponds they may disturb fish and contribute to stress responses.

9.8.4 Mitigation measures

Recent studies have shifted away from viewing grebes as problem species to fisheries and instead investigate them as indicators of ecosystem condition. These studies reinforce that fish availability strongly influences grebe breeding success, making the species a useful bioindicator of freshwater ecosystem health rather than a primary threat to fisheries. As a consequence most management is orientated around coexistence rather than predator control, including restoring structurally diverse fish habitats and using pond netting where local conflicts occur.

The species is listed on Appendix I of the Convention on Migratory Species, and is covered by the African Eurasian Waterbird Agreement. It is listed under Annex II of the Bern Convention and Annex I of the EU Birds Directive. Nevertheless actions are taken to manage great crested grebe populations, largely for protection of other flora and fauna. These largely oriented around disturbance and destruction of eggs or capture and keeping, the latter exclusively in France (Figure 29). Disturbance methods were found to be largely ineffective in a controlled replicate experiment trying to deter grebes from a commercially important bleak (*Alburnus alburnus*) fishery (Gagliardi et al., 2006). Killing is rarely carried out and was mainly restricted to The Netherlands and Poland.

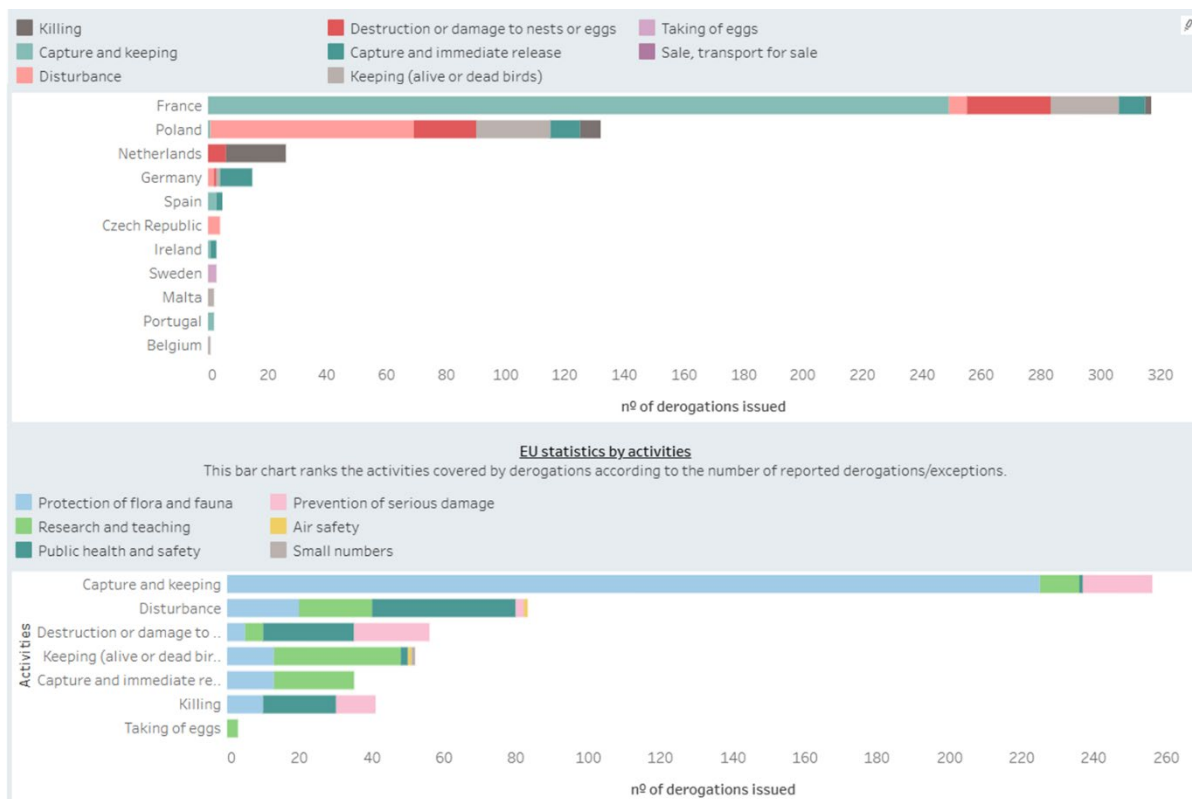


Figure 29. Number of derogations issued under Article 9 of the Birds Directive to control great crested grebe in Europe: 2020–2025. Source: European Environment Agency (2024a) Overview of derogations and exceptions to species protection across the EU. <https://www.eea.europa.eu/en/analysis/maps-and-charts/overview-of-derogations-and-exceptions-dashboards>. Cited 07 August 2026

In conclusion predation by great crested grebes has considerably smaller ecological and economic impacts on freshwater fisheries and aquaculture than cormorants or mergansers. Their predation is largely focused on abundant small fish, and there is little evidence that they cause widespread declines in fish stocks or significant economic losses. The most productive management strategies therefore combine habitat restoration, fisheries monitoring, and targeted non-lethal mitigation where local conflicts arise, rather than broad-scale population control

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9.9 Dalmatian pelican *Pelecanus crispus*

9.9.1 Distribution and abundance

The Dalmatian pelican *Pelecanus crispus* has two populations in Europe, one in the Black Sea and Eastern Mediterranean and a second population east of the Black Sea across Asia. The European population of *Pelecanus crispus* is estimated at a minimum of 5 693-6 694 breeding pairs, and potentially as high as 7 342-8 984 pairs (Catsadorakis, 2017) converting to population estimate of 11 400-13 400 mature individuals (BirdLife International, 2021; <https://datazone.birdlife.org/search>). *Pelecanus crispus* is listed as near threatened and declining on the IUCN Red List (BirdLife International, 2021).



Figure 30. Distribution of Dalmatian pelican in Europe highlighting location of resident ●, breeding ● and non breeding ● areas and discrimination of discrete populations (dotted line) (Source: <https://criticalsites.wetlands.org/en>)

9.9.2 Habitat and ecology

The Dalmatian pelican is one of Europe's largest freshwater-associated birds, with a high food requirement. It occurs mainly at inland, freshwater wetlands, but also at coastal lagoons, river deltas and estuaries (Crivelli et al., 1997; Mix and Bräunlich, 2000; Peja et al. 1996; del Hoyo et al., 1992; Elliott et al., 2017), breeding on small islands in freshwater lakes or in dense aquatic vegetation (Crivelli, 1994; Peja et al., 1996; Pyrovetsi, 1997; del Hoyo et al., 1992).

It often occurs in large flocks in winter, foraging communally and cooperatively in small groups (Cramp and Simmons, 1977). The birds return to their breeding sites in late-January to April, depending on the region (Nelson, 2005). Immature birds and non-breeders may remain in the wintering grounds year

round, or may stay with the breeding colonies (Cramp and Simmons, 1977; Crivelli et al., 1998; Nelson, 2005).

The species is exclusively piscivorous, exploiting a wide range of fish species, including carp *Cyprinus carpio*, perch *Perca fluviatilis*, rudd *scardinius erythrophthalmus*, roach *Rutilus*, and pike *Esox lucius* in freshwater wetlands, and eels, mullet, gobies and shrimps in brackish waters (Crivelli 1994; del Hoyo et al. 1992; Peja et al. 1996). They can consume approximately 1.2 kg of fish per bird per day (rewildingbritain.org.uk²³).

9.9.3 Ecological and economic impact of predation

There is evidence that pelican population dynamics are linked with the availability of local fish resources, indicating pelicans are directly selecting abundant prey and can regulate fish availability (Pyrovetsi and Economidis, 1998). This interpretation must, however, be treated with caution because fish populations are also regulated by other sources of natural mortality, including predation by other animals, disease, fishing pressure and environmental conditions (Figure 2), and such an interpretation has been forwarded by Shumka et al. (2023) for the decline in fish yield in the Karavasta lagoon, (south-eastern Adriatic Sea). Nevertheless increasing numbers of Dalmatian pelicans in Greece has resulted in considerable conflict between fisheries and bird conservation, although an increase in great cormorant numbers in the region may be contributing to the decline in fisheries, e.g. Prespa lake system (Catsadorakis et al. 2024).

There is currently little evidence to conclude that Dalmatian pelicans are a major source of economic losses across the European aquaculture sector. Extensive carp and nursery ponds could be vulnerable because they provide shallow ponds, dense fish populations, predictable food and relatively easy access.

There is little information on the economic impact of Dalmatian pelicans on fisheries and aquaculture, but consumption of up to 1.2 kg/day per bird would equate to over 2500 tonnes consumed per year (based on a median 12,000 mature individuals and occupancy for 180 days per year). This would represent a considerable proportion of commercial catches in regional fisheries, such as seen on Mikri Prespa in Greece where pelicans consume large volumes of the endangered endemic fish species *Alburnus belvica*, and create a conservation conflict between the birds and fish (Pyrovetsi and Economidis, 1998).

9.9.4 Mitigation measures

The Dalmatian pelican is listed on Appendix I of the Convention on Migratory Species, and is covered by the African Eurasian Waterbird Agreement. It is listed under Annex II of the Bern Convention and Annex I of the EU Birds Directive. Consequently any actions in the EU must align with these agreements and tend to be orientated around coexistence rather than predator control where local conflicts occur. Nevertheless derogations have been granted (Figure 31), although the data are somewhat confusing since France and Poland have the greatest number of application, but the species is not present in any significant numbers in either country.

²³ https://www.rewildingbritain.org.uk/why-rewild/reintroductions-key-species/key-species/dalmatian-pelican?utm_source=chatgpt.com

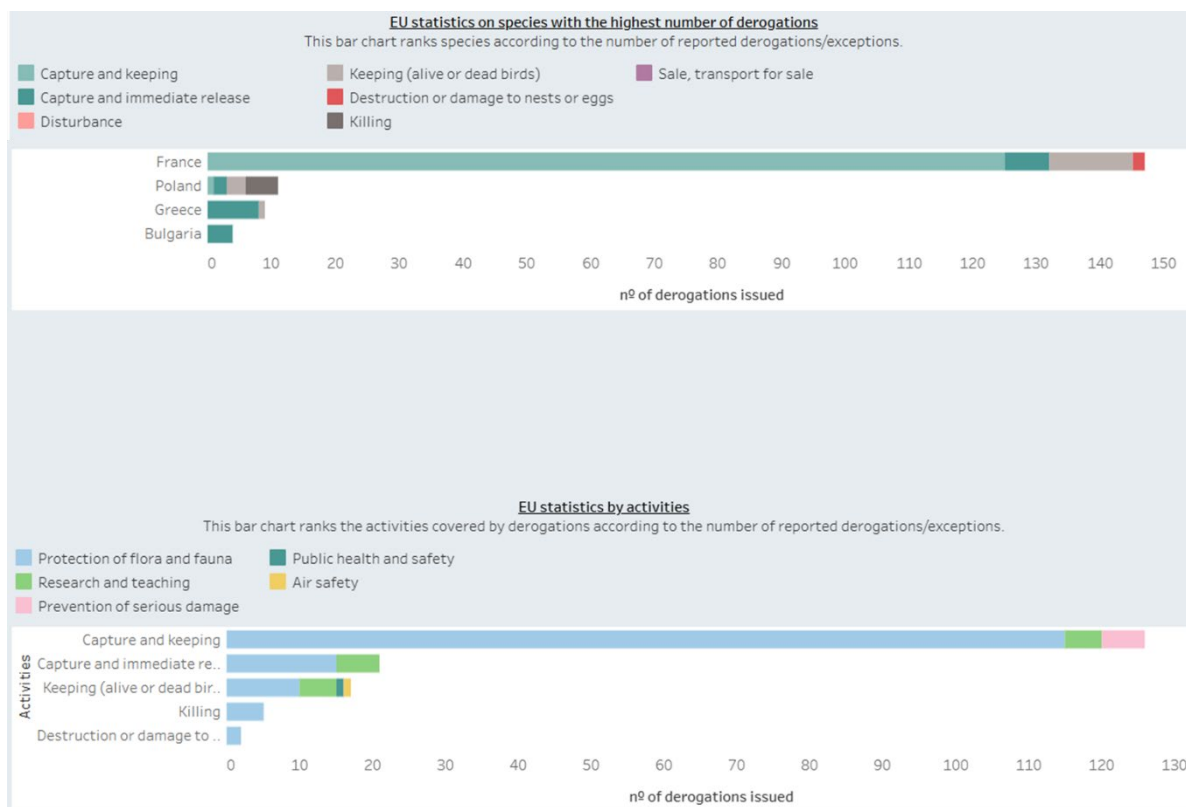


Figure 31. Number of derogations issued under Article 9 of the Birds Directive to control Dalmatian pelican in Europe: 2020–2025. Source: European Environment Agency (2024a) Overview of derogations and exceptions to species protection across the EU. <https://www.eea.europa.eu/en/analysis/maps-and-charts/overview-of-derogations-and-exceptions-dashboards>. Cited 07 August 2026

There is little information about potential measures to mitigate problems arising from Dalmatian pelican predation on fisheries and aquaculture systems. Installing a series of horizontal wires spaced at intervals over aquaculture ponds has been highlighted as a measure to prevent the species from depredating farmed fish (Crivelli, 1996), but most Article 9 derogations seem to be associated with capture rather than lethal or scarring and disturbance techniques (Figure 31). There also seems to be an AEWA Single Species Action Plan prepared for the Dalmatian pelican (Catsadorakis and Portolou, 2018; Crivelli, 1996) but this is not ratified.

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9.10 Great white pelican - *Pelecanus onocrotalus*

9.10.1 Distribution and abundance

Great white pelican - *Pelecanus onocrotalus* – is found in Europe from the Eastern Mediterranean the Black Sea and across the Caucasus and Central Asia (Figure 32). The European population of *Pelecanus onocrotalus* is estimated as 4 900-5 600 pairs, which equates to 9 700-11 100 mature individuals (BirdLife International, 2015). *Pelecanus onocrotalus* is listed as Least Concern on the IUCN Red List and the overall population trend in Europe are considered is increasing (BirdLife International, 2021).



Figure 32. Distribution of great white pelicans in Europe highlighting location of resident ●, breeding ● and non breeding ● areas and discrimination of discrete populations (dotted line) (Source: <https://criticalsites.wetlands.org/en>)

9.10.2 Habitat and ecology

Great white pelicans are associated with relatively large open fresh and brackish water bodies, (del Hoyo et al. 1992; Johnsgard 1993; Snow and Perrins 1998). The species requires areas of extensive reedbeds, wet swamps, mudflats and sandbanks or gravel and rocky substrates for nesting (del Hoyo et al. 1992; Johnsgard 1993; Nelson 2005; Snow and Perrins 1998).

Northern populations are fully migratory (Figure 33) and travel via stop-over sites, typically large lakes, between breeding and feeding areas (del Hoyo et al. 1992; Nelson 2005). The species breeds in the spring in large colonies of 200 to 40,000 pairs (Brown et al. 1982; Snow and Perrins 1998; Nelson 2005) (occasionally with other species such as Dalmatian pelican *Pelecanus crispus*) (Flint et al. 1984).

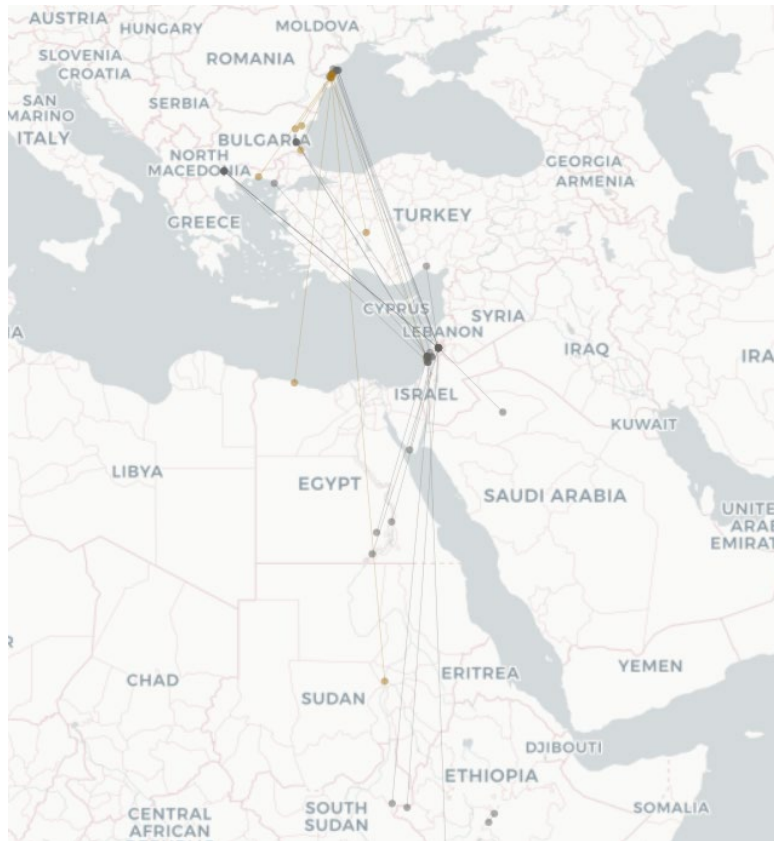


Figure 33. Migration patterns of great white pelican from The Eurasian African Bird Migration Atlas Source: Bird migration atlas: great cormorant - <https://migrationatlas.org/index.php/node/1706>)

The species is exclusively piscivorous, capturing fish that can generally be swallowed whole, with prey size constrained by the dimensions of its bill and gape. This has important implications for fisheries because great white pelicans can consume both juvenile and marketable-sized fish, unlike smaller piscivorous birds whose predation is often largely restricted to fry and small juveniles. Great white pelicans frequently forage cooperatively, typically in groups of 8–12 individuals (Brown et al., 1982; del Hoyo et al., 1992), driving fish into shallow water where they can be captured efficiently. They may also travel considerable distances between breeding or roosting colonies and feeding areas, with large flocks of 50–500 birds reported (del Hoyo et al., 1992; Snow & Perrins, 1998). Foraging activity is concentrated primarily during the early morning and evening (Johnsgard, 1993), potentially resulting in substantial localised predation pressure where large numbers of pelicans aggregate at productive fishing sites.

9.10.3 Ecological and economic impact of predation

The great white pelican is a substantial fish predator, and there is considerable conflict with fisheries and aquaculture. They can consume 0.8 and 1.1 kg of fish per bird per day (Guillet and Furness, 1985; Shmueli et al., 2000). Natural feeding sites for pelicans have diminished during recent decades due to human activities, and sites of extensive aquaculture have become the favoured feeding places for wintering and migrating great white pelicans. Farms may have little bird activity for most of the year and then experience substantial problems when large migratory flocks arrive. Fish farms install deterrents such as nets and scaring devices that raise the cost of production. The fisheries sector has reported considerable impact on fish and yields, and conflict between pelicans and fishers has escalated in recent years resulting in pelicans being shot.

The scale of the impact is seen from the one record of predation impact reported in the EIFAAC/EEA/FEAP (2026). Some 14 200 birds were reported inhabiting Romanian water bodies for about 90 days in each of 2023 and 2024 causing an estimated EUR 7 033 000 each year. Just the total number of birds raises questions about the European population size which is estimated at around 10 400 mature individuals across its range. Nevertheless, the volume of fish potentially consumed by the estimated 10 400 mature adults inhabiting European waters for 90 days based on a daily intake of 1 kg/day would be substantial at 936 tonnes.

9.10.4 Mitigation measures

The great white pelican is listed on Appendix I of the Convention on Migratory Species, and are covered by the African Eurasian Waterbird Agreement. It is listed under Annex II of the Bern Convention and Annex I of the EU Birds Directive. Consequently any actions in the EU must align with these agreements and tend to be orientated around coexistence rather than predator control where local conflicts occur. Nevertheless derogations have been granted against the species ,although the data are somewhat confusing as France and Poland have the greatest number of application, given the species is not present in either country.

There is little information about potential measures to mitigate problems arising from great white pelican predation on fisheries and aquaculture systems. Installing a series of horizontal wirses spaced at intervals over aquaculture ponds is the main measure to prevent the species from depredating farmed fish (Crivelli et al. 1991), and is in line with most Article 9 derogations that are associated with capture rather than lethal, and scarring and disturbance techniques (Figure 34).

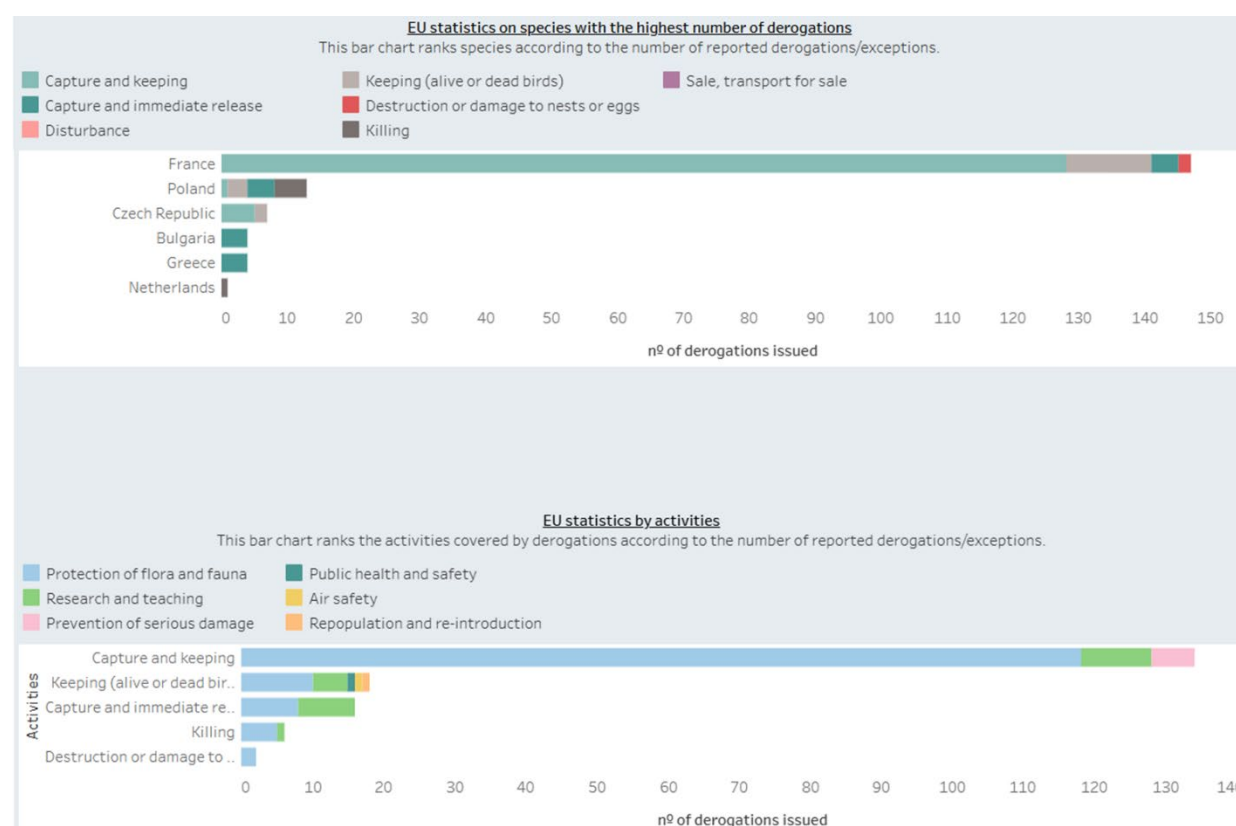


Figure 34. Number of derogations issued under Article 9 of the Birds Directive to control Dalmatian pelican in Europe: 2020–2025. Source: European Environment Agency (2024a) Overview of derogations and exceptions to species protection across the EU. <https://www.eea.europa.eu/en/analysis/maps-and-charts/overview-of-derogations-and-exceptions-dashboards>. Cited 07 August 2026

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9.11 Eurasian otter - *Lutra lutra*

9.11.1 Distribution and abundance

The Eurasian otter is a native semi-aquatic predator of European freshwater ecosystems (Loy et al., 2026). The species experienced major historical declines because of persecution, hunting, pollution, habitat loss, wetland drainage and deterioration of riparian habitat. It has subsequently recovered in much of Europe since the 1990s following legal protection, especially under the Habitats Directive, and environmental improvements, (Figure 35) (Chanin, 2003), although pressures from agriculture, invasive species and land-use change remain prominent.

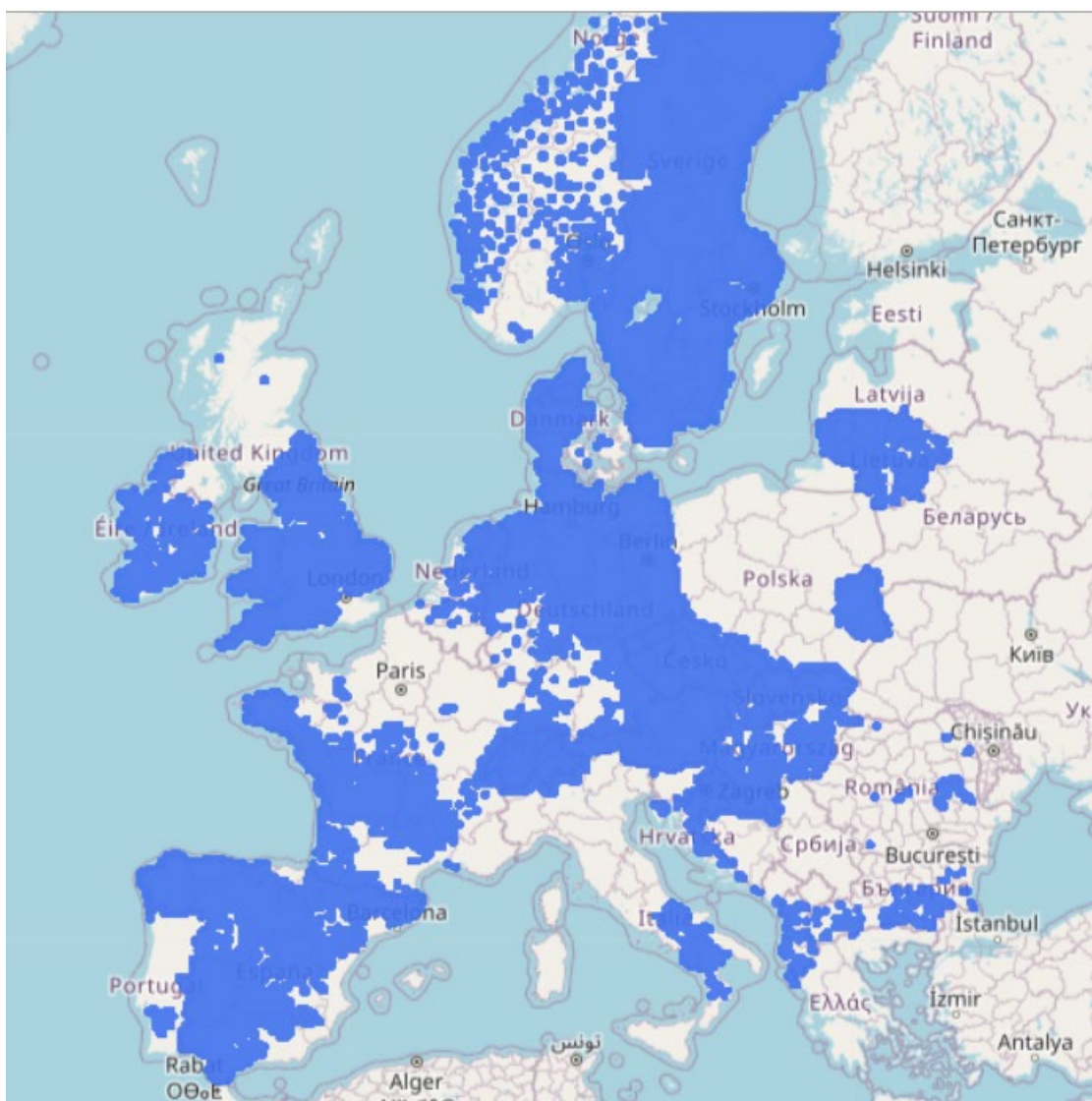


Figure 35. Distribution of European otter (source: OtterSpotter <https://www.otterspotter.de/en/verbreitung#resultanchor>)

The European Environment Agency currently lists *Lutra lutra* as Near Threatened at the European level, while the species is protected under the EU Habitats Directive, including Annex II and Annex IV. More than 4,000 Natura 2000 sites are designated for the species.

9.11.2 Habitat and ecology

The Eurasian otter is predominantly piscivorous, with fish comprising approximately 60–90% of dietary biomass in freshwater habitats (Beja, 1996; Britton et al., 2017; Drake et al., 2023; Krawczyk et al., 2016). Otters tend to prey on abundant, slow-moving, or benthic fish species, such as eels (*Anguilla anguilla*), cyprinids (e.g., roach and bream), and small percids (Clavero et al., 2003). Larger or more agile species are less frequently captured, reflecting constraints imposed by prey behaviour and capture success. Otters typically consume approximately 10–15% of their body weight daily, making consistent prey access essential (Chanin, 2003). Their foraging behaviour is influenced by factors such as water depth, flow conditions, and habitat structure, which affect prey accessibility. Although fish dominate the diet, the species remains opportunistic and may consume amphibians and crustaceans when available (Britton et al., 2017; Drake et al., 2023; Krawczyk et al., 2016). This adaptability allows them to persist in a wide range of habitats, from upland streams to lowland rivers and lakes.

Foraging is typically nocturnal in areas with high human disturbance, which allows them to exploit fisheries and aquaculture sites with reduced disturbance (Kruuk, 2006; Tolrà et al., 2024a). Their hunting efficiency is enhanced by highly sensitive vibrissae that detect prey movement in turbid water. This tactile hunting adaptation allows them to forage effectively even in poor visibility conditions.

As native predators, they help regulate fish populations, especially weaker or more accessible individuals, and contribute to ecosystem balance and biodiversity (Britton et al., 2017). The presence of otters is considered a sign of healthy ecosystem because their presence reflects good water quality and sufficient prey availability. Their recovery across Europe is therefore often interpreted as evidence of improving environmental conditions (Tolrà et al., 2024b; Roos et al., 2015).

9.11.3 Ecological and economic impact of predation

The Eurasian otter is a generalist predator with a strong preference for fish, but the impact of otter predation on wild fish populations is a subject of debate (e.g. Václavíková et al., 2011). Most studies conclude that otters exert limited long-term effects on healthy wild fish populations, particularly in larger river systems where prey abundance remains high (Clavero et al., 2003). Their effects are density-dependent and compensatory, rather than leading to long-term declines in fish stocks (Kruuk, 2006; Clavero et al., 2003).

Otter predation can conflict with conservation goals where they prey upon economically valuable or protected fish species, such as salmonids (Carss et al., 1990; Sortland et al., 2023). This interaction is complex, however, as otters may also contribute to the control of invasive or overabundant species, depending on the ecosystem context (Kloskowski, 2011). Isolating the effects of otter predation from other factors, such as habitat degradation, overfishing, or pollution, is therefore challenging: the ecological impact of otter predation is highly context-dependent and cannot be generalized across all freshwater systems. As a result, attributing fish population declines solely to otters is often problematic and may oversimplify complex ecological processes (Carss et al., 1990; Cianfrani et al. 2011).

Nevertheless, localised impacts may occur in small streams or isolated water bodies where fish populations are naturally limited (Carss, 2003). In such habitats, repeated predation may temporarily reduce juvenile recruitment or alter species composition, especially where habitat fragmentation restricts recolonisation (Krawczyk, et al., 2016). Greater problems or conflicts arise when numbers of otters increase and spillover into stocked fisheries and aquaculture systems.

Otter predation can have pronounced effects in small, enclosed systems such as fish ponds, artificial systems or heavily managed fisheries. High local densities of otters, combined with limited prey refuges, may lead to reductions in fish stocks, particularly for commercially valuable species (Kloskowski, 2005). Stocked fish, particularly species such as trout and carp, often lack predator awareness and are more susceptible to predation. Additionally, fish stocking can create unnaturally high prey densities, which may attract otters and increase their local population density. This can lead to repeated predation events by individual otters, intensifying local ecological impacts (Sittenthaler et al. 2015). In some cases, otters may preferentially target larger fish, including breeding individuals, which can disproportionately affect population dynamics and fishery sustainability (Kortan et al., 2007).

The expansion of otter populations in Europe has created increasing conflict with freshwater aquaculture (Poledníková et al., 2013). Fish farms, trout ponds, carp ponds, and hatcheries often contain high densities of fish within confined spaces, making them attractive feeding sites for otters (Kloskowski, 2005). The concentration of fish in ponds or enclosures provides predictable and abundant food resources. Unlike in natural systems, captive fish have reduced opportunities to evade predators, which can increase predation efficiency.

The most immediate economic impact of otter predation is loss of fish biomass. Fish farms and recreational fisheries frequently report reductions in stock due to otter activity, particularly in high-density aquaculture systems (Kloskowski, 2011). In recreational fisheries, the loss of large, high-value specimen fish can affect angler satisfaction and reduce revenue from fishing sales. These economic losses can vary substantially depending on type of fishery, farm design, geographic location and local otter density, but losses can be considerable (Table 7).

Table 7. Regional overview of estimate annual damage by otter on fish populations in reporting European countries (source FAO, 2026b)

Country	Otter population (individuals)	Occurrence in the year (days)	Estimated damages caused (EUR, average annual)	Reference years
Upper Austria and Styria	936	365	1 817 500	2023-2024
Croatia	30	365	58 254	2023-2024
Czechia	3322	345	6 008 400	2023-2024
Finland	4000	365	7 767 200	2023-2024
Republic of Moldova	800	360	1 532 000	2023-2024
Netherlands	450	365	873 810	2023
Poland	2248	311	3 719 361	2023-2024
Romania	1782	365	3 391 500	2023-2024
Total	13 343	355	25 168 025	2023-2024

The impact on aquaculture occurs in several forms:

- Direct consumption - Otters consume marketable fish directly, particularly larger individuals with high commercial value. Salmonids and carp are frequently targeted because of their size and accessibility.

- Injury to stock - Otters may injure more fish than they consume. Bite wounds can lead to infection, poor growth, and later mortality among surviving stock (Lanszki et al., 2001). The problem can be escalated because otters sometimes over kill and leave dead and dying fish behind.
- Stress-related losses - The presence of predators can induce chronic stress in farmed fish. Repeated disturbance may suppress feeding, reduce growth rates and increase susceptibility to disease.
- Infrastructure damage - Otters may damage fencing, nets or pond banks while attempting to access fish stocks, creating additional financial burdens for aquaculture operators (Kloskowski, 2011).

Although total biomass losses at a regional scale may be relatively modest, the economic impact at individual sites can be substantial. This is particularly true for small-scale aquaculture operators, who may lack the resources to absorb repeated losses. Reduced fish stocks in recreational fisheries can also lead to lower catch rates, diminishing their attractiveness to anglers and resulting in lost income. These opportunity costs are difficult to quantify but are an important component of the overall economic impact.

In addition to direct losses, fisheries incur a range of indirect costs associated with mitigating otter predation. These include investments in exclusion measures such as fencing, netting, and electric barriers, as well as ongoing maintenance and monitoring expenses. Restocking costs can also be substantial, particularly where predation pressure is persistent.

A key characteristic of otter-related economic impacts is their uneven spatial distribution. While some fisheries experience severe losses, others are largely unaffected. This variability depends on factors such as habitat structure, fish species composition, stocking practices and the availability of natural prey. Small, enclosed, or intensively managed systems are particularly vulnerable, whereas large, natural river systems tend to be more resilient. Consequently, the economic burden of otter predation is often concentrated among specific groups, leading to perceptions of disproportionate impact. These situations give rise to conflicts between conservation objectives and economic interests.

9.11.4 Mitigation measures

The management of otter predation is complicated by legal protections. The Eurasian otter is strictly protected under European legislation, including the Habitats Directive, which limits the use of lethal control measures (Roos et al., 2015). Although derogations are granted in certain circumstances (Figure 36), these are tightly regulated. As a result, fisheries must rely primarily on non-lethal mitigation strategies, which can be costly and not always fully effective (Stepien et al., 2024). These include:

- predator-proof fencing
- electric exclusion barriers
- improved pond design
- habitat management to reduce farm accessibility
- covered raceways for hatcheries
- controlled stocking schedules
- compensation schemes for verified losses.

Fencing off fish farms and stocked ponds can be effective but requires continuous maintenance as otters are able to bypass fences with time (Allen et al., 2020). These approaches can reduce conflict while

maintaining conservation commitments, but the challenge lies in balancing species recovery with economic interests (Kloskowski, 2011). This requires a combination of ecological understanding, stakeholder engagement, and policy support. Successful management will depend on cooperation between conservation authorities and aquaculture producers and fishery interests, and mechanisms to compensate the aquaculture and fisheries owners.

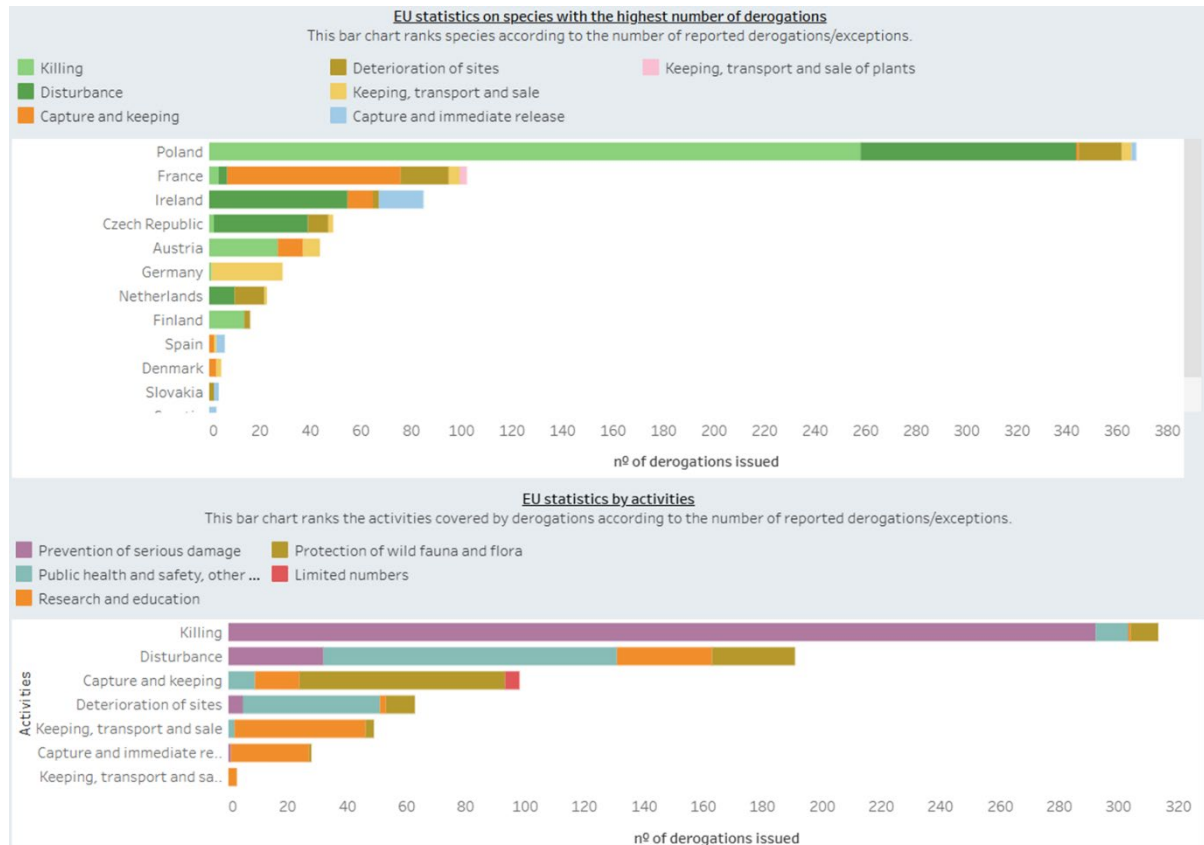


Figure 36. Number of derogations issued under Article 12 of the Habitats Directive to control otters in Europe: 2020–2024. Source: European Environment Agency (2024a) Overview of derogations and exceptions to species protection across the EU. <https://www.eea.europa.eu/en/analysis/maps-and-charts/overview-of-derogations-and-exceptions-dashboards>. Cited 07 August 2026

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9.12 American mink - *Neogale vison* syn. *Mustela*/*Neovision vison*

9.12.1 Distribution and abundance

The American mink was introduced into Europe primarily through the fur-farming industry during the twentieth century. Feral populations subsequently became established in many European countries and it is now one of Europe's most widespread invasive carnivores, with a distribution covering most of the continent except for some parts of southern and south-eastern Europe. Its range is still increasing, driven by high adaptability and historical fur-farm escapes (Bonesi and Palazon, 2007; Vada et al., 2023). The European Union has now formally recognised the American mink as an invasive alien species of Union concern. The European Commission added *Neogale vison* to the list of species of concern under Regulation (EU) 2025/1422, following assessment under the EU invasive alien species regulation.

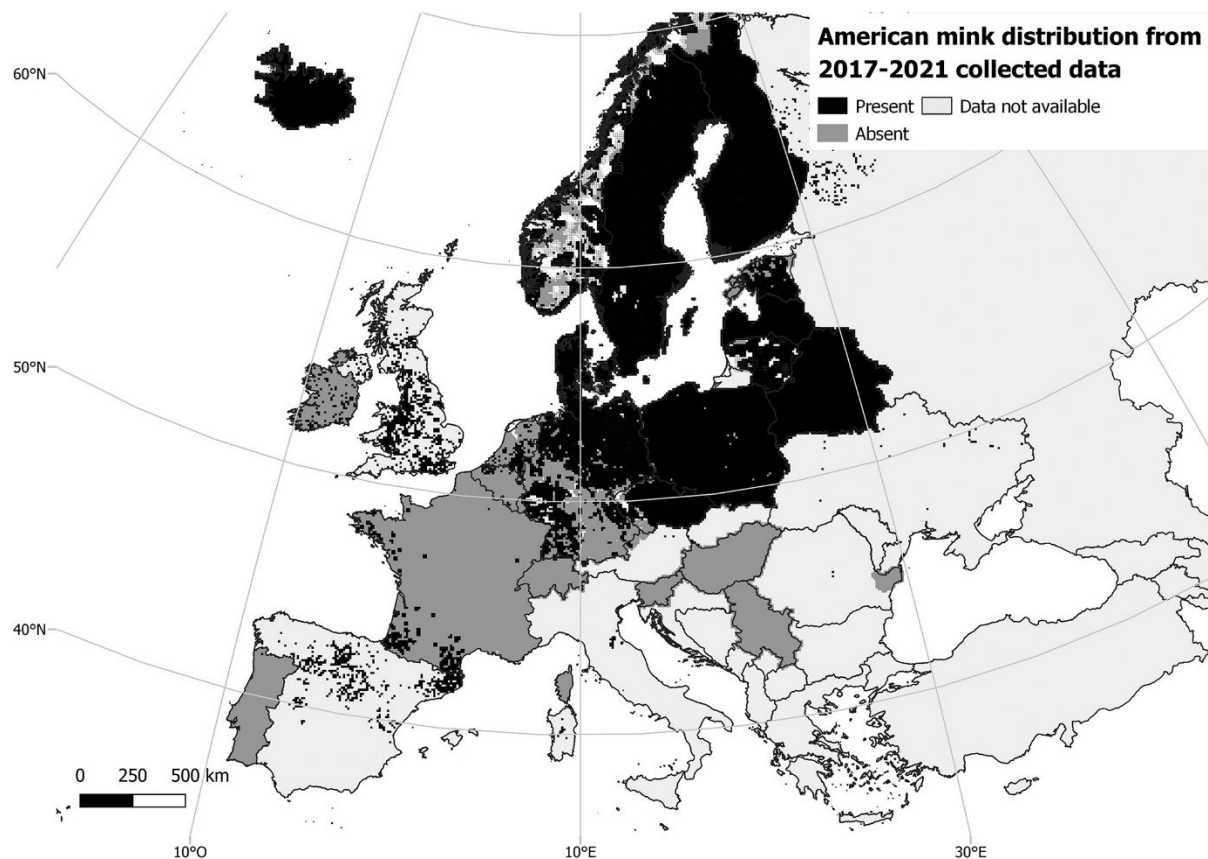


Figure 37 Distribution of American mink in Europe 2017-2021. (Source: Vada et al., 2023)

9.12.2 Habitat and ecology

The American mink is a highly opportunistic predator adjusting its diet to local prey, with a diet dominated by rodents, fish, birds, and amphibians (Lanszki et al., 2003; Bonesi et al., 2004; Krawczyk et al., 2013; Zschille et al., 2014; Chibowski et al., 2019). In agricultural landscapes, rodents, especially voles, often dominate the diet (Bonesi et al., 2004). The feeding habits and trophic niche overlap between sympatric otter and mink have been studied extensively (e.g. Bonesi et al., 2004). The otter is known to specialize in aquatic prey, while mink can exploit both aquatic and terrestrial prey. Otters outcompete mink for aquatic prey, leading mink to seek out terrestrial prey and

undergo a feeding niche shift to a higher proportion of mammals and birds, particularly in winter. Predation tends to be spatially patchy and influenced by habitat (rivers vs lakes vs estuaries).

9.12.3 Ecological and economic impact of predation

The mink species causing most of the fisheries conflict in Europe is the American mink, *Neogale vison*. It should not be confused with the European mink, *Mustela lutreola*. American mink is recognised as an invasive alien carnivore with biodiversity impacts, especially on fish and amphibians.

Across Europe, economic damage to fish farms from American mink is generally small at the national scale, but locally it can be significant, especially where mink densities are high or farms lack adequate protection (Moore et al., 2000). Fish farms, especially open-pen salmonid systems or pond-based carp farms, provide easy access to fish, limited escape opportunities for prey and structures that mink can use for concealment. This makes fish farms attractive for mink, particularly in regions with high feral populations (e.g. UK, Iceland, parts of Scandinavia).

Economic losses include predation on stocked fish, stress-related mortality and damage to nets or infrastructure. Damage levels vary widely because they depend on local mink density, farm design and protection, proximity to suitable mink habitat and availability of alternative prey. In the Outer Hebrides of Scotland, for example, mink have caused considerable damage to salmon farming interests.

Mink can inflict damage to native biodiversity, poultry runs, reared game birds and fisheries (Harrison and Symes, 1989; Moore et al., 2000; Sheail, 2004). They have been found to impact native water voles (Woodroffe et al., 1990), ground nesting sea birds (Craik, 1997), and waterfowl (Ferrerias and Macdonald, 1999). Their depredation, including surplus killing, has been linked to almost complete breeding failure amongst colonies of terns and gulls, including some rare species, which can have repercussions for the eco-tourist industry (Craik, 1997). American mink can also have considerable impact on salmonid juveniles, including stocked fish (Moore et al., 2000; Lanszki et al. 2003). Vehanen et al. (2022) also found that hatchery-reared trout were more vulnerable to mink predation than their wild counterparts. The effect was associated with behavioural differences between hatchery and wild fish and larger stocked fish being more vulnerable to predation.

9.12.4 Mitigation measures

The inclusion of American mink as an invasive alien species of Union concern in Regulation (EU) 2025/1422 is an important development for fisheries and aquaculture. It provides a stronger policy basis for prevention, containment and eradication. Countries differ strongly in their American mink management objectives. Many have projects aimed at eradication or control, either at a national or local level that have been successful (Roy et al. 2009, Léger et al. 2018, Martin and Lea 2020), but others are less proactive. In the UK, prevention of economic damage by trapping and restricting access appears to be satisfactory for most types of damage caused by and American mink (Harrison and Symes, 1989). Fencing off fish farm ponds and stocked fisheries seems to be an option used where feasible, although this brings high costs for installation. Adding woody material to increase habitat heterogeneity and reduced predation by mink is also an option (Vehanen et al. 2022). This means habitat restoration may simultaneously improve fish production and reduce mink predation.

One notable success is eradication of American mink from a large area of eastern England after undertaking a comprehensive trapping exercise operated by volunteers under professional supervision (Martin and Baker 2026). Mink were all but eliminated after 3 years of coordinated

trapping. This highlights the need for coordinated control policies to align management objectives across countries, which are currently lacking (Santulli et al. 2014).

American mink is an important freshwater fish predator, particularly in small salmonid streams where juvenile fish are concentrated and vulnerable. However, there is insufficient evidence to conclude that mink are a major Europe-wide driver of freshwater fish declines or that they impose a large Europe-wide economic cost on fisheries and aquaculture. Their effects are strongly site, habitat, season and prey-dependent.

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9.13 Seals

9.13.1 Species of interest

There are two main species of seal found in European waters – the grey seal (*Halichoerus grypus*) and harbour or common seal (*Phoca vitulina*). In addition, there are two species with restricted distribution in the Baltic Sea – the Baltic ringed seal (*Phoca/Pusa hispida botnica*) and Saimaa ringed seal (*Phoca/Pusa hispida saimensis*).

9.13.2 Distribution and abundance

Atlantic grey seals and harbour seals are found across the North-East Atlantic Ocean (Figure 38 and Figure 39). Their numbers have increased in recent years following protective legislation and designation of Natura 2000 sites. Distribution of the Baltic ringed seal is exclusively in the north part of the Baltic Sea in the Bothnia Bay, Gulf of Riga and Gulf of Finland, and the Saimaa ringed seal exclusively in Lake Saimaa, Finland.

All species are protected under the European Union Habitats Directive (Council Directive 92/43/EEC) Annex II and V and the Conservation of Migratory Species of Wild. EU Member States must also designate and manage Natura 2000 sites for seals to contribute to the achievement /maintenance of favourable conservation status. Currently, the grey seal is in favourable condition status across much of its range, except Denmark, Germany and Poland, but harbour seals are mostly in unfavourable status (HELCOM, 2023). The Baltic ringed seal and Saimaa ringed seal are listed as unfavourable status and the Saimaa ringed seal is listed as Endangered on the IUCN Red List. Both grey and harbour seal and the ringed seal (*Pusa hispida*) are listed as Least Concern by the European Commission Red List for Europe (HELCOM, 2023).

Seal abundance and distribution are influenced by many factors, such as disease, pollutants, competition with other species, changes in the distribution and abundance of prey, disturbance, and anthropogenic removals such as by-catch, hunting and culling. Seals were hunted well into the 20th Century, resulting in depleted populations. They are still vulnerable to bycatch in fishing gears, such as commercial trawls, gill nets and fixed nets.

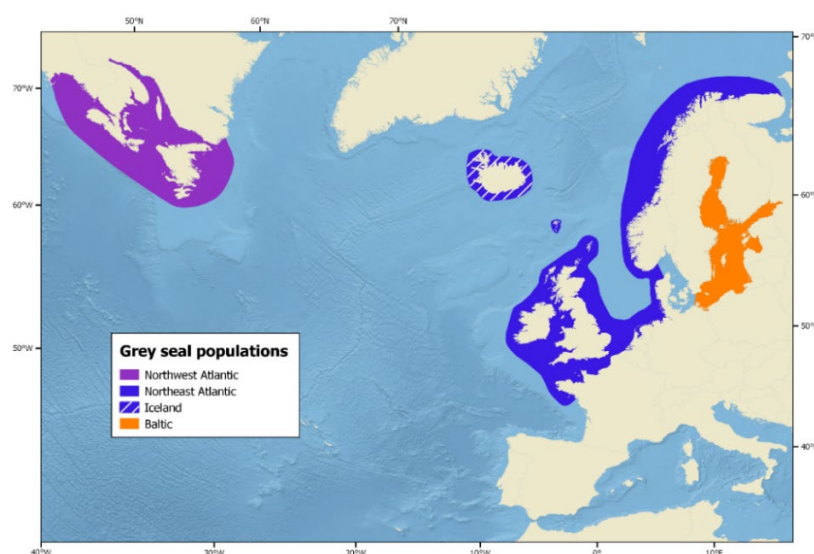


Figure 38. Distribution of grey seal in Europe highlighting location of discrete populations (Source: <https://nammco.no/grey-seal/>)

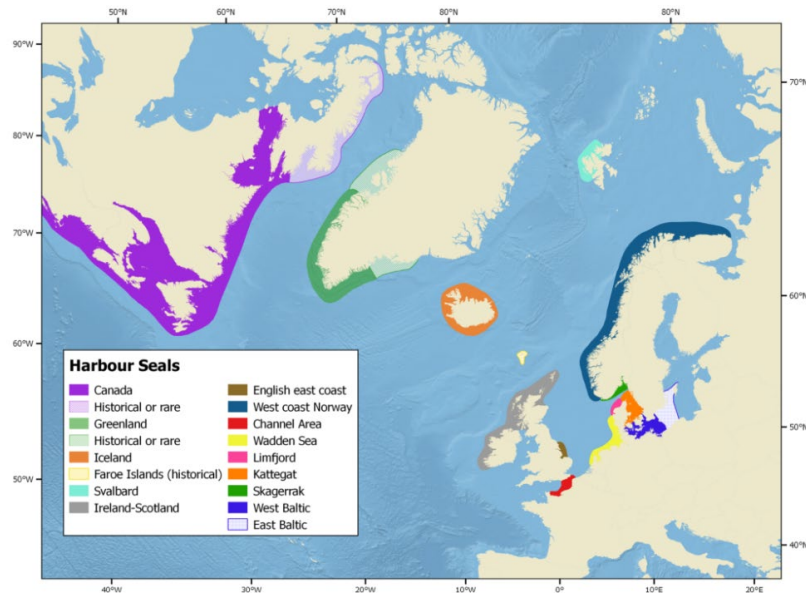


Figure 39. Distribution of harbour seal in Europe highlighting location of discrete populations (Source: <https://nammco.no/harbour-seal/>)



Figure 40. Distribution of ringed seal in Europe highlighting location of discrete populations (Source: <https://nammco.no/ringed-seal/>)

Information on abundance of each species is variable but over 250 000 grey seal, 120 000 harbour seal, 10 000 Baltic ringed sea, and less than 300 Saimaa ringed seal are reported (NAMMCO²⁴. Banga et al., 2022)

9.13.3 Habitat and ecology

Seals are adapted to temperate and cold environments and, in Europe, are primarily associated with the North Atlantic Ocean, North Sea and Baltic Sea. Grey and harbour seals forage in coastal and offshore waters, but regularly return to land-based haul-out sites to rest, moult and breed. Individual

²⁴ <https://nammco.no/seals-and-walrus/>

seals may use these sites throughout the year or during specific periods associated with breeding or moulting. Harbour seals generally forage within ≈ 50 km of their preferred haul-out sites, whereas grey seals are capable of travelling much farther offshore, often more than 100 km from land, and may use multiple haul-out sites throughout the year. Grey seals and harbour seals overlap in spatial distribution and prey resources, potentially resulting in competition of food, which could result in the seals seeking alternative food resources and in some cases exploiting estuarine and riverine resources (Gilles, 2009; Damseaux et al., 2021; Planque et al., 2021)

Grey and harbour seals are opportunistic feeders and consume a range of prey depending on region and availability (Gilles, 2009; Lundström et al., 2010). Both species are apex predators in marine and coastal ecosystems, but their ecological influence extends into freshwater environments through their use of estuaries and rivers. The diet of seals in European waters is diverse and includes a range of commercially and ecologically important fish species, such as eel, hake, whiting, cod, haddock, pollock and flatfish, as well as crustaceans and cephalopods such as shrimps and squid and octopus. In estuarine and riverine environments, seals exploit species that migrate between marine and freshwater habitats, including Atlantic salmon (*Salmo salar*), sea trout (*Salmo trutta*) and European eel (*Anguilla anguilla*) (Thompson et al., 2007). They need to eat 4-6% of their body weight in food each day.

Unlike predators restricted to freshwater habitats, seals are highly mobile and can exploit prey resources across marine, estuarine and freshwater environments. Their movements and foraging activity are influenced by prey availability, tidal cycles and river-flow conditions. This flexibility allows them to switch between marine, estuarine and freshwater prey according to local availability (Smout et al., 2014). Both harbour and grey seals regularly enter rivers to forage, particularly in systems supporting migratory fish such as salmonids (Thompson et al., 2007). These incursions are often associated with predictable concentrations of prey, including salmon migrations and spawning aggregations, rather than occurring randomly.

Seals may also concentrate their foraging activity around fish passes, dams and other structures that restrict or delay fish movements. Such locations can create predictable concentrations of prey and consequently increase their vulnerability to predation. Individual seals have been recorded travelling considerable distances upstream and, in some cases, repeatedly exploiting particular river systems, especially where migratory fish occur at high densities (Thompson et al., 2007). Consequently, although seal movements between marine and freshwater environments represent an important natural ecological linkage, repeated use of particular rivers can result in spatially concentrated predation pressure and create management concerns for fisheries and fish conservation.

The composition of seal diets varies both geographically and seasonally. Predation on salmonids, for example, is likely to increase during periods of migration when fish become concentrated and are therefore more accessible to predators. Seals foraging farther upstream may also consume resident freshwater fish in addition to migratory species. Although seals may preferentially target energetically profitable prey, including larger fish when these are readily available, their feeding behaviour is fundamentally opportunistic. They are capable of exploiting a broad range of prey species and adjusting their diet according to local and seasonal prey availability.

9.13.4 Ecological and economic impact of predation

Although seals are primarily marine mammals, their role as predators of inland and coastal fish in Europe has received increasing attention. Certain seal species, particularly the harbour seal and grey seal, frequently forage in inshore, estuarine and riverine environments, where they conflict with

inshore fisheries and specifically prey on migratory and resident freshwater fish (Aarts et al., 2019). Their expanding populations and behavioural adaptation to inland foraging have generated management concern (e.g. Thompson et al. 2007, 2019). In particular, seal predation can contribute to declines in vulnerable fish populations, particularly where stocks are already under pressure from other factors such as overfishing, habitat degradation and climate change (Wilson and Hammond, 2019). Predation near river mouths and migration bottlenecks may be especially critical.

However, other perspectives²⁵ emphasise the difficulty of quantifying these impacts and caution against attributing fish population declines solely to seal predation. A major limitation is the difficulty of isolating predation effects from other drivers of fish population decline, including overfishing, habitat degradation, barriers to migration and climate change. As with other predators, the effects of seals are context-dependent and influenced by a range of ecological and environmental factors (Hansen and Hardin, 2006; Thompson et al., 2019), but there is substantial evidence that seals can consume large amounts of fish and compete with other predators and fisheries (Hansson et al., 2018; Svendsen et al. 2025). For example, Tuus²⁶ estimated that grey seals consume 12 000 t of fish annually in Estonian water and this is comparable to the 12 000 t landed by coastal fishing and 12 000 t consumed by cormorants. Damage to coastal fisheries was already estimated at EUR 1million in 2009 when 3900 seals were present, equivalent to 20-30% of the coastal fisheries revenue¹¹. Today more than 7000 seals are present.

Beyond this direct depredation, seals also cause considerable damage to fish caught in nets making them unmarketable reducing income for fishers, and this has resulted in cessation of some fishing operations and reduction in the fishing fleets, especially around the Baltic Sea (Svendsen et al. 2025). Similarly seals can cause considerable damage to coastal aquaculture enterprises. Here they cause considerable damage to aquaculture cage netting to access fish, resulting in direct loss through consumption or injury to individual fish and escape of stock through damaged net cages.

9.13.5 Mitigation measures

Seal predation on inland fish has led to increasing conflict with fisheries, particularly in regions where economically important species such as salmon are affected. Seals are protected under national and international legislation, including the EU Habitats Directive, which complicates management interventions. Balancing conservation objectives with fisheries interests remains a key challenge. Nevertheless, legal removal of seals to protect fisheries or for hunting is still carried out in some European countries, and the threat from by-catch remains present across many areas.

In some European countries, including the UK and Scotland, management responses have included monitoring programmes, non-lethal deterrents and, in certain cases, licensed culling (Thompson et al., 2019). Indeed there are many reported derogations under Article 12 of the Habitats Directive to control numbers of seal. (Figure 41). But there are also mechanisms available to reduce the interaction between seals and fisheries. These include modification of fishing traps design (Suuronen et al. 2006; Ljungberg et al. 2022) and acoustic deterrents (Lehtonen et al., 2022; Lehtonen and Petri Suuronen, 2025; Suuronen et al., 2023). Indeed there are suggestions to use acoustic deterrents to create seal-free fishing areas (Lehtonen et al., 2023) and the methodology has been adapted to work in rivers (Veneranta et al., 2024), but how practical this would be for all the salmon and sea trout rivers in

²⁵ Scientists: Wrong to link seals and cormorants to overall decline of Baltic herring - Stockholms universite

²⁶ <https://www.nsrac.org/wp-content/uploads/2025/02/7HERKI1.pdf>

Europe is questionable. In addition, some compensation is available for damaged gear but this is insignificant (EUR 20 000)

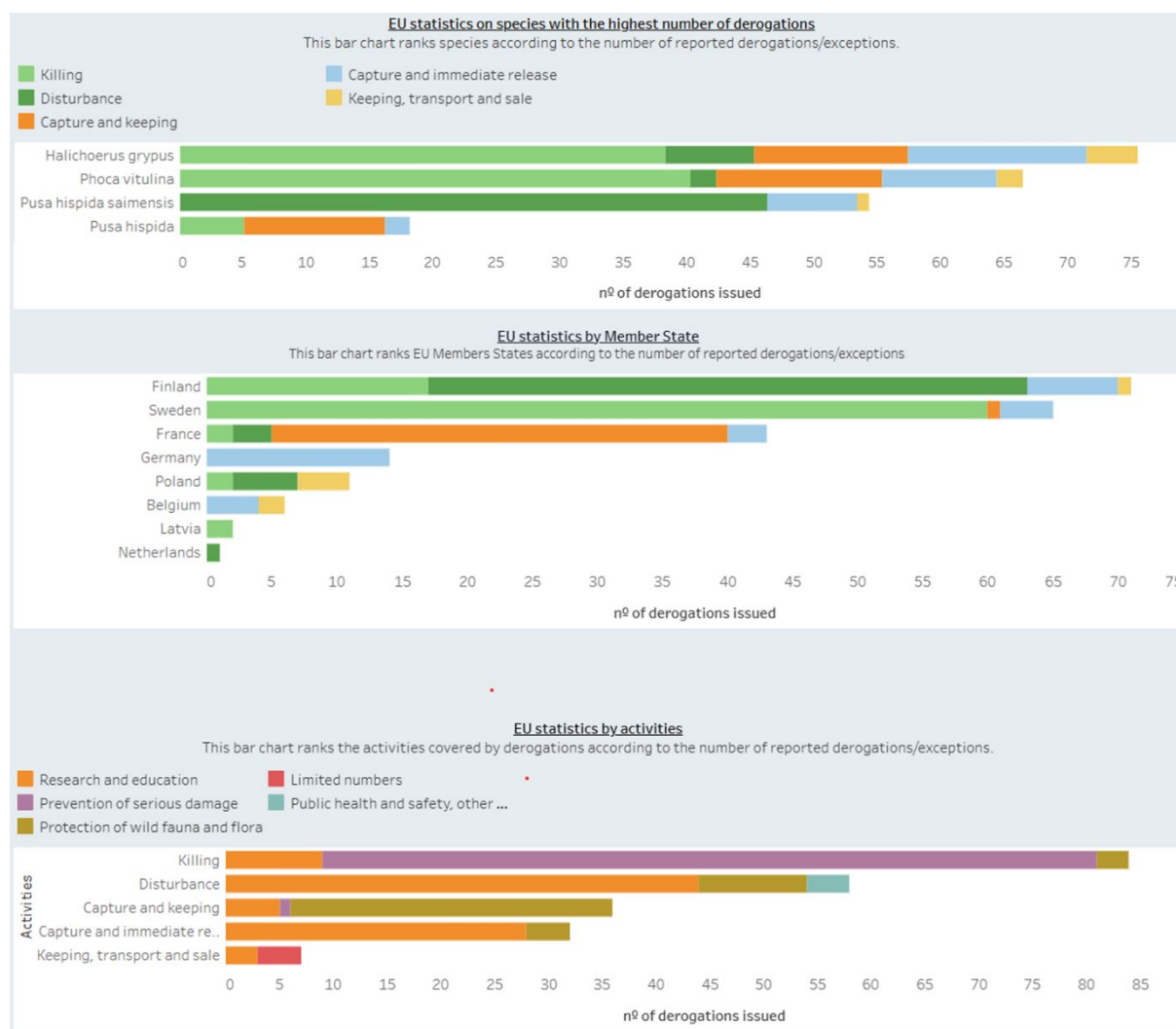


Figure 41. Number of derogations issued under Article 12 of the Habitats Directive to control seals in Europe: 2020–2024. Source: European Environment Agency (2024a) Overview of derogations and exceptions to species protection across the EU. <https://www.eea.europa.eu/en/analysis/maps-and-charts/overview-of-derogations-and-exceptions-dashboards>. Cited 07 August 2026

Another option being proposed is modifying the EU ban on trade of seal products. Currently seal products can only be utilized within the hunter's household. Moderating this use would avoid dumping legally caught seals at sea and rational use of the products (Suuronen et al., 2026).

Seals are significant predators of inland fish in Europe, particularly in estuarine and riverine systems where marine and freshwater environments intersect. Several key debates persist with the management of seals. One major issue is the extent to which seal predation affects fish populations, particularly migratory species of conservation concern. While some suggest significant impacts, others argue that these are often overstated or confounded by other pressures. While seal predation may be locally important, especially at migration bottlenecks, its broader ecological effects remain difficult to quantify. There is no doubt that there is considerable economic damage of seal predation, especially on coastal fisheries and aquaculture, and, thus, there is a need for more integrated, ecosystem-based studies that consider the cumulative effects of multiple stressors on fish populations, including predation, environmental change and human activities to help resolve the conflict. Future work should

prioritise long-term, interdisciplinary approaches that integrate ecological, behavioural and socio-economic perspectives to provide a more balanced and robust understanding of these complex interactions.

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9.14 Pike - *Esox lucius* L.

9.14.1 Ecology and distribution

Pike are widely distributed around the northern hemisphere (Figure 42) (Craig, 1996). Only one species, *Esox lucius*, is indigenous to European waters. It is found throughout Europe with the exception of Iceland, southern Italy, the Iberian Peninsula, and southern Balkans, although it has been introduced to the Iberian Peninsula (Elvira et al., 1996; Godinho et al., 1997; Elvira, 1998).



Figure 42. Native distribution of pike (*Esox lucius*) across the Northern Hemisphere

Pike are typically found in lakes, slow-flowing rivers and canals with shallow to moderately deep waters and a high density of vegetation (Raaijmakers, 1988; Grimm, 1994). Young pike are usually associated with weed beds in shallow, sheltered areas (Bregazzi and Kennedy, 1980; Raaijmakers, 1988). By contrast, adult pike are often found in clear waters with a substrate of silt, clay, peat, or sand (Frost and Kipling, 1967; Grimm, 1989). Those found inhabiting deeper, more turbid waters tend to be the larger individuals. Pike are non-territorial fish, their distribution is often related to that of their prey (Bregazzi and Kennedy, 1980; Casselman and Lewis, 1996; Minns, et al., 1996; Eklov, 1997). Movements are generally to shallow areas in the spring (spawning migration; Lucas, 1992), and to deeper parts over winter (Raaijmakers, 1988). Pike follow a daily pattern of activity associated with feeding at dawn and dusk when their visual acuity is better than that of their prey (Beukema, 1970).

Pike are non-selective predators, and abundance and seasonal availability of food organisms determines food selection (Frost, 1954; Vostradovsky, 1981; Chapman et al., 1989). Size-selectivity of prey has been demonstrated (Frost, 1954; Mann, 1976). Pike usually taking food of about a third of their own length, although prey items of up to 70 per cent of body length have been recorded. When foraging, pike will remain concealed among vegetation or other available cover until prey is located. Initially this is by visual stimuli, followed by precise location using the lateral organ.

The feeding activity of pike follows a seasonal pattern related temperature, prey abundance, and reproduction (Frost, 1954; Lawler, 1965). Feeding activity is greatest during May, immediately after spawning, and remains high throughout the summer and early autumn (Bregazzi and Kennedy, 1980), but decreases over winter (Allen, 1939). Pike undergo a fast associated with the spawning period between March and April (Frost, 1954).

In the first few months of life, the diet comprises zooplankton (particularly Cladocera and Copepoda) and chironomid larvae. At a length of 20+ mm pike develop tendencies towards piscivory. The exact timing of this piscivory is dependent on the availability of prey fish to the young pike. The size of the prey eaten increases with increasing size of the individuals. Hence, as the fish grow, the diet changes to include larger prey items such as *Asellus*, *Gammarus*, smaller perch fry, minnows and sticklebacks. Pike >200 mm feed predominantly on fish, mainly perch, roach, rudd, gudgeon, bream, eels, trout and smaller pike (Toner and Lawler, 1969). Large invertebrates and vertebrates, (birds, mammals and amphibians) may also be eaten, although rarely. Cannibalism can occur as soon as there is a sufficient size differentiation between juveniles, and increases in conditions of food shortage or overcrowding. Juveniles that resort to cannibalism in such instances have faster growth than their non-cannibalistic siblings (Giles et al., 1986.).

Predation and cannibalism are also important in regulating the number of recruits to the population; cannibalism being considered a density-dependent regulator (Kipling and Frost, 1970; Grimm, 1981; Mann, 1982; Raat, 1990). The biomasses of 0+, <350 mm, and <410 mm pike are negatively correlated with those of larger pike, mainly as a result of cannibalism (Grimm, 1994). This relationship tends to increase as other suitable food items become scarce. Different size groups of pike usually inhabit different areas, but if food is limited the distance over which the fish will forage increases, increasing their exposure to intraspecific predation.

9.14.2 Ecological and economic impacts

There are numerous examples of impact of pike introduced into water bodies outside its natural range, especially from North America (e.g. DeBates et al., 2008; Nicholson et al., 2015; Sepulveda et al., 2013; Spens and Ball, 2008). There are also examples of the impact of pike introductions on native species in Europe. For example, Byström et al. (2007) demonstrated that the invasion of northern pike into a small subarctic lake caused major changes in the fish community through top-down predation effects. Pike replaced Arctic charr as the dominant predator and reduced populations of nine-spined stickleback by an order of magnitude. The study suggested that the superior piscivory of pike, combined with competition for shared prey resources, contributed to the exclusion or local extirpation of Arctic charr from the lake.

Similarly, Hesthagen et al. (2014) documented the severe impact of introduced pike on a brown trout population in a small Norwegian lake system. Within four to five years of pike establishment, the previously abundant brown trout population had virtually disappeared, with all age classes declining and age 2+ trout experiencing the greatest reductions. In the absence of major environmental changes, the collapse of the trout population was attributed primarily to predation by pike following their colonisation of the lake and its feeder streams.

By contrast, there is considerable information on the beneficial role of pike in biomanipulation initiatives (Mehner et al., 2004; Sndergaard et al., 1997). Essentially the pike population size is manipulated to ensure that heavy predation pressure regulates the numbers of small planktivorous and benthivorous fish (Klinge et al., 1995; Goldyn et al., 1997). They are frequently stocked to predate on the juvenile planktivorous and benthivorous cyprinids and control their numbers (Grimm and Backyx, 1994; Klinge et al., 1995; Sndergaard et al., 1997; Mehner et al., 2004). This action is only appropriate where pike is endemic and should not be considered in water bodies outside its natural range.

The economic perspectives surrounding pike all orientate towards the value of pike in fisheries and the benefits of stocking pike to create recreational fisheries (Arlinghaus et al., 2020, 2023; Mickiewicz

and Trella, 2019). There is no available study regarding the impact of pike on other fisheries other than the cost of control where it has been introduced and become invasive (e.g. Schwörer et al., 2012).

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9.15 Pikeperch - *Sander lucioperca* (L.)

9.15.1 Ecology and distribution

Pikeperch is a percid species native to northern and central Europe (Freyhof and Kottelat, 2008) (Figure 43). It is now spreading and becoming more common throughout Western Europe as a consequence of numerous introductions (Hickley, 1986; Smith et al., 1998). Pikeperch is typically found in lowland rivers and large still waters (Deelder and Willemsen, 1964). Its ideal habitat is shallow, eutrophic, turbid and well-oxygenated water (Hickley, 1986; Lammens et al., 1992), and a hard substrate (Wheeler, 1969). In rivers, pikeperch are found in slow-flowing reaches where there is little turbulence (Deelder and Willemsen, 1964).



Figure 43. Distribution map of *S. lucioperca* (Freyhof and Kottelat, 2008). Mustard denotes native populations, purple is introduced populations.

Pikeperch have limited with home ranges typically less than 1 km, and potentially explains the slow colonisation from places of release in new water bodies. (Hickley, 1986, Koed et al., 2002; Aarts and Breukelaar, 2017; Nolan et al., 2024)

Pikeperch are efficient predators, capturing their prey in a similar way to pike. They hunt prey in open water, on the bottom and among vegetation (Deelder and Willemsen, 1964; Kitchell et al., 1977; Craig, 1987). They usually feed in conditions of low light intensity, to which their sight is well adapted by the development of the *tapetum lucidum*, a reflective layer within the retina of the eye. Daily feeding rates can change significantly with light intensity (Fedorova and Drozhzhina, 1982).

Pikeperch form shoals as young fish while feeding on plankton, and become solitary with age (Craig, 1987; Van Densen, 1985). Larvae begin to feed within 48 hr of hatching, at a size of ≈ 5 -6 mm (Linfield and Rickards, 1979). Fry are pelagic and live on zooplankton for the first few months of life (Kokes, 1993; Frankiewicz et al., 1997). Typical prey includes copepods, cladocerans, occasionally rotifers, chironomids and other insect larvae (Wheeler, 1969; Craig, 1987). At a size of ≈ 25 mm pikeperch become piscivorous, although fish remain a relatively unimportant part of the diet until they reach a length of 100 mm (Buijse and Houthuijsen, 1992; Collette et al., 1977; Ginter et al. 2011; Hansson et al., 1997). At this size, zander become predominantly piscivorous, although a proportion of non-fish food items, particularly invertebrates, is consumed (Bousseba et al., 2024; Linfield and Rickards, 1979).

Their main prey consists of shoaling fish species such as roach, ruffe, perch and their own young (Biro, 1973; Craig, 1987), but they exploit the full range of forage fish available; the diet changing with differences in availability and abundance of prey (Craig, 1987; Peltonen et al., 1996).

Pikeperch have a relatively small gape compared with pike, hence the maximum size of prey that can be caught and handled is correspondingly smaller (Deedler and Willemsen, 1964; Linfield and Rickards, 1979; Kell, 1985; Dörner et al., 2007). As with perch, the extent of cannibalism depends on the abundance of young pikeperch compared to that of other prey fish species (Puke, 1952). Hence, in a similar way to perch, the intensity of cannibalism increases when there is a lack of other available food (Raat, 1990; Mehner et al., 1996).

9.15.2 Ecological and economic impacts

Pikeperch has been introduced across Europe but the effects of these introduction remains unpredictable, with differing impacts in different countries (Van Densen, 1994; Fickling and Lee, 1983). For example in Turkey, the UK, The Netherlands where introductions have caused a continuum from little long-term impact to extirpation of species (Crivelli, 1995; Linfield, 1984) such as perch, roach and small bream all declining following the release of pikeperch into a Dutch lake (Lammens et al., 1992), as did the populations of roach, perch, bream and pike in Scandinavia (Jepsen, et al., 2000). Although pikeperch and cyprinids can coexist in eutrophic and turbid waters (Leah and Kell, 1985), negative impacts on stocks of small fish species often coincide with the introduction of pikeperch (van Densen and Grimm, 1988; Lammens et al., 1992). Nevertheless, after many years of stabilization, the prey populations in some areas appear to recover after the initial decline (Adams, 1987; Hickley, 1986; Smith et al., 1997).

One other dimension that must be considered when assessing the impact of introduced pikeperch is the behavioural response of their prey (Diehl and Eklöv, 1995). They evoke an immediate local change in abundance and species composition of prey fish by influencing immigration and emigration (He and Kitchell, 1990). Some fish may avoid predation by hiding in dense littoral vegetation beds or in the oxygen depleted and predator-free hypolimnion, where present (Persson, 1991a, b; Braband and Faafeng, 1993). Such predator avoidance can prevent the utilisation of feeding grounds and increase inter- and intraspecific competition (Persson, 1983), which in turn may lead to an alteration in fish community structure (Barthalemes, 1988). Ultimately that can impact vulnerability of fish to angling.

Finally, while pikeperch may be considered detrimental to recreational and sport fisheries, it does support valuable commercial fisheries, mainly in lakes (Lehtonen et al., 1996), and aquaculture production (Ruuhijärvi and Hyvärinen, 1996). However, there is considerable debate (Löffler, 1998) about whether stocks have to be supported by stocking to maintain yield (Ruuhijarvi et al., 1996; Hanson et al., 1997a).

Predation by piscivorous fishes is in general not a problem in most water bodies where the predators are in balance with the ecosystem dynamics and predator prey relationships under those dynamics. Where this is disrupted by introduction of non-native piscivorous fishes such as pikeperch this can be a problem requiring control of the predators by targeted culling of caught fish, systematic culling through targeted sampling or leaving the system to stabilise. Whilst the control mechanism can reduce the predator population they all meet with problems, particularly where anglers are reluctant to destroy the fish (Smith et al., 1997; Nolan et al. 2019) or in the case of stabilisation, this may take many years and not achieve the desired outcome (Linfield, 1984).

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9.16 Perch - *Perca fluviatilis*

9.16.1 Ecology and distribution

Perch, *Perca fluviatilis*, is widely distributed throughout Eurasia and has been introduced into Spain and Portugal (Ning et al., 2025; Thorpe, 1977a, b, c). Perch is predominantly a freshwater species inhabiting lowland ponds or lakes and slow to moderately flowing rivers and streams, but may also be found in brackish waters.

Percids are favoured in moderately productive systems, particularly where there is a high proportion of submerged macrophytes (Persson, 1991, 1993). The animal food on which perch predate is associated with littoral vegetation, and perch foraging efficiency increases under these conditions. Furthermore, the abundance of perch also increases with more complex habitat (Eklov, 1997) and is a good indicator of habitat quality (Wolter and Vilcinskis, 1997a, b). As the productivity of the habitat increases, the amount of submerged vegetation decreases due to lower light penetration. The dominance of percids gives way to cyprinids, which are better able to utilise non-animal food resources, in highly productive conditions (Persson, 1991).

Perch are typically gregarious in nature, with shoals often uniform with respect to the size of fish (Jellyman, 1980). Older perch may become solitary (Bruylants et al., 1986). They exhibit a diurnal activity pattern, with maximum activity around dawn and dusk, and very little movement at night (Guma'a, 1982; Slavik, 1992). The increased activity at low light intensities has been suggested to be an advantage to perch because they are able to maintain food foraging dominance over other species (Craig, 1977). The subsequent decrease in light penetration, and therefore water clarity, enables the perch to feed for longer during the day during the summer, a time which is the optimum period for growth.

Perch are generally opportunistic feeders, eating a wide variety of organisms throughout their life. They are efficient foragers especially in dense, submerged vegetation (Diehl, 1988). There is a tendency for perch to feed in shoals, and solitary fish are less able to capture prey fish species (Deelder, 1951). Perch tend to exhibit seasonal variation in feeding intensity, with maximum food intake from June to September and a partial fast over winter (Collette et al., 1977; Craig, 1987).

The principal factor governing the feeding behaviour of larval perch is the availability of suitable food organisms (Treasurer, 1992; Mehner et al., 1996). Water temperature also appears to play a critical role, as the development and abundance of zooplankton populations are directly related to temperature. This is particularly significant for 0+ perch, which, unlike other species such as pike, are dependent on zooplankton throughout the first summer of life (Guma'a, 1978).

Copepods, rotifers, and cladocerans form the dominant food items for larval perch. However, when zooplankton is lacking, the diet is supplemented with benthic organisms such as leeches, ephemeropteran nymphs, chironomid larvae and *Gammarus* (Bregazzi and Kennedy, 1982). Perch usually become piscivorous as they approach a length of approximately 140 mm (Deelder, 1951; Craig, 1974, 1987), although piscivory has been described at smaller sizes (Craig, 1987).

Adult perch tend to eat a wider variety of food organisms than their juvenile counterparts; the young perch being restricted by the size of prey they are able to handle. The diet includes items such as Odonata, *Asellus*, ostracods, beetle larvae, most species of small fish and fry of large species such as perch and roach. Bregazzi and Kennedy (1982) found that fish under 150 mm ate mainly benthic organisms and some fish, whereas those over 150 mm were predominantly piscivorous.

9.16.2 Ecological and economic Impacts

There is little evidence of perch impacting wild fish populations except perhaps where they have been introduced (Buchaca et al., 2011; Hayden et al., 2013; Eloranta et al., 2016; Ning et al., 2025). Several studies undertaken in the subarctic zone of Europe have shown that the trophic generalist behaviour of translocated European perch has led to them having negative impacts on other fish species, such as whitefish (*Coregonus lavaretus*) (Hayden et al., 2013; Eloranta et al., 2016). In Lake Skadar (located on the border between Montenegro and Albania), the European perch has been the most dominant species in relative biomass within the fish community, even 35 years after its first appearance in the lake (Mrdak et al. 2018). Similarly, in Lake Furnas (located on the Azores Archipelago, Portugal), the introduction of European perch (in 1913) has led to significant declines in the populations of carp and roach (also introduced and stocked in 1890 and 1895, respectively) (Gonçalves, 1997; Buchaca et al., 2011).

Conversely, intense cannibalism is common in perch populations. It is probably a density-dependent mechanism, caused by the high availability of perch young compared with young of other species (Holcik, 1977). Consequently perch population density is controlled according to the food resources available (van Densen et al., 1996; Mann, 1978). Further, where food resources are limited, stunted growth has been reported (Persson, 1983b; Persson and Greenberg, 1990). This stunting in perch populations is likely caused by a juvenile “competitive bottleneck” with other species competing for larger zooplankton at early life stages (Burrough et al., 1979; Kennedy et al., 1983; Thorpe, 1974) or simply overcrowding and/or competition from an abundance of small fish Williams (1967).

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9.17 European catfish - *Silurus glanis*

9.17.1 Ecology and distribution

The European catfish *Silurus glanis*, also called wels catfish or sheatfish, is a large species of catfish native to wide areas of central, southern, and eastern Europe, in the basins of the Baltic, Black and Caspian seas (Verreycken, 2019) (Figure 44). While native to parts of Eastern Europe, its widespread introduction across Western and Southern Europe as a sport fish and for aquaculture has generated increasing concern regarding its ecological and economic impacts (Copp et al., 2009; Verreycken, 2019).

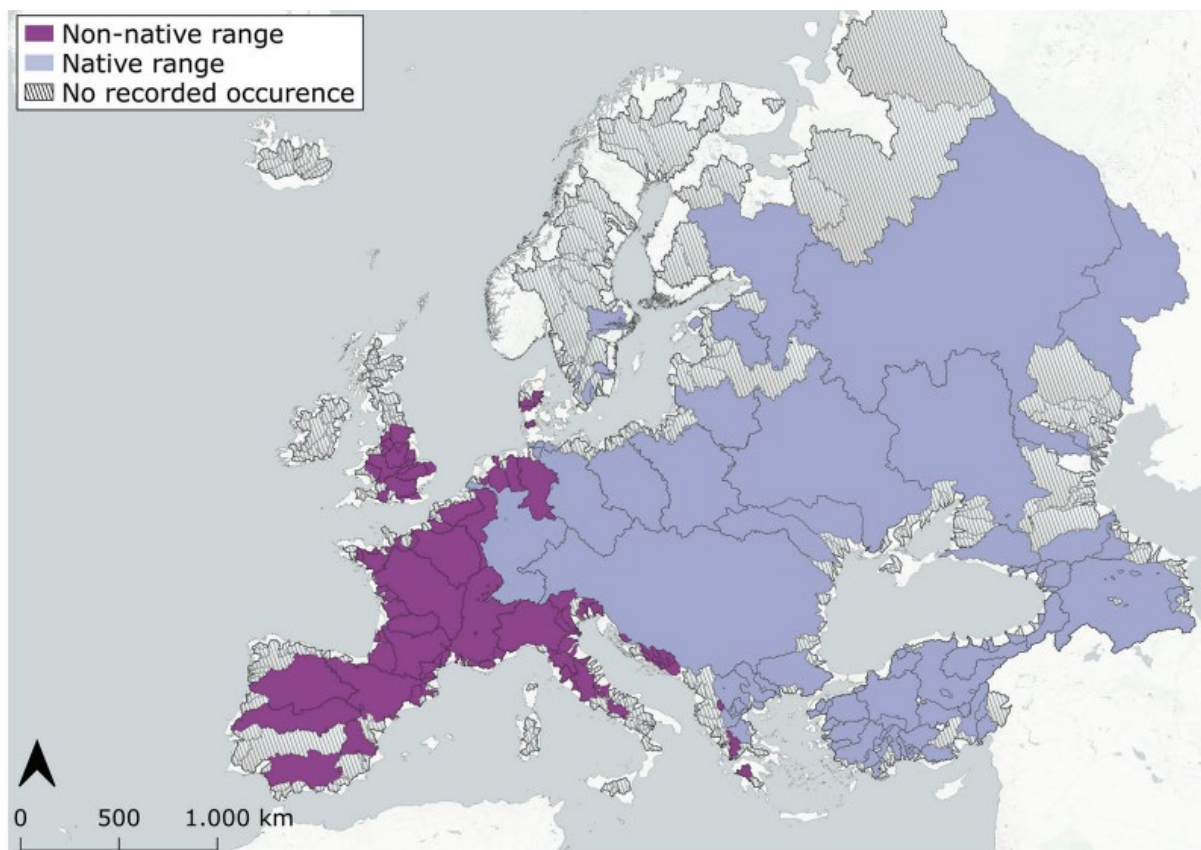


Figure 44. Native (blue), introduced (purple) and unconfirmed (grey striped) distributional ranges of European catfish (*Silurus glanis*) in Eurasian watersheds (source: Brignone et al. 2026)

European catfish inhabit slow-flowing, deep lowland rivers and adjacent still waters such as lagoons, marshes and lakes. It is largely nocturnal, although in deep, clouded water it may be active on dull days during daytime, but mostly it feeds at sunset and just before dawn. The European catfish is an apex predator, reaching lengths of over 2 m (Alp et al., 2011; Horoszewicz and Backiel, 2012) and occupying a wide range of freshwater habitats, including rivers, lakes and reservoirs. The European catfish is an opportunistic predator with a broad diet, capable of consuming a wide range of prey across trophic levels (Copp et al., 2009; Syväranta et al., 2009; Boulêtreau et al., 2011; Vejřík et al., 2017; Boulêtreau et al., 2018, 2020, 2021; Říha et al., 2025). Evidence suggests that the species exerts strong top-down control on fish communities, alters trophic dynamics, and can contribute to biodiversity loss in non-native systems (Cucherousset et al., 2012, 2018). This trophic flexibility allows it to exploit a wide range of prey and maintain high biomass in invaded ecosystems (Castaldelli et al., 2013; Cucherousset et al., 2018).

9.17.2 Ecological and economic impacts

Ecological impacts of *S. glanis* appear to vary across ecosystems (Copp et al., 2009), ranging from negligible or marginal (Guillerault et al., 2015) to severe (Castaldelli et al., 2013; Encina et al., 2024; Santos et al., 2024), but its introduction has been associated with declines in native fish populations (Verreycken, 2019). Consequently, the species is often considered invasive in Western and Southern Europe (Copp et al., 2009; Brignone et al., 2026). Its presence can lead to homogenisation of fish communities and loss of endemic species, particularly in Mediterranean river systems (Castaldelli et al., 2013; Carol et al., 2009). For example, after the establishment of *S. glanis* in the lowland backwaters of the Po River in north eastern Italy, some native species (*Alburnus arborella* and *Scardinius erythrophthalmus*) declined in abundance and biomass or disappeared (*Rutilus aula* and *Tinca tinca*) (Castaldelli et al., 2013). Also long-term studies in Iberian rivers have shown reductions in the abundance and biomass of native cyprinids following the establishment of wels catfish (Boulêtreau et al. 2023; Ferreira et al. 2019). Since its introduction in the Mequinenza Reservoir in 1974, it has spread to other parts of the Ebro basin, including its tributaries, especially the Segre River. Some endemic Cyprinidae species of Iberian barbels, genus *Barbus* that were once abundant, especially in the Ebro River, have disappeared due to competition with and predation by wels catfish (Carol et al., 2009). The ecology of the river has also changed, with a major growth in aquatic vegetation such as algae, potentially destabilizing the ecological balance. Such impacts are particularly pronounced in systems with endemic species that evolved without large piscivorous predators. The limited and often contradictory evidence on the impacts of *S. glanis* is reflected in its EICAT classification, where it has been assigned a “MR – MAJOR” rating for predation on native communities, but with low confidence due to the largely descriptive nature of the evidence and the prevalence of Data Deficient sources.

The economic impacts of European catfish on inland fisheries are complex and vary between systems. On one hand, predation on commercially valuable fish species can reduce yields and affect fishery profitability. In Spain for example, establishment of European catfish has led to declines in native fish populations, particularly Iberian barbel, with measurable reductions in abundance and biomass. This reduction is also notable in systems where fisheries rely on stocking programmes, as stocked fish may be heavily preyed upon (Boulêtreau et al., 2023). On the other hand, the European catfish itself is a valuable resource. It supports commercial fisheries in some regions of Eastern Europe and Russian Federation, and is a highly prized species in recreational angling, particularly for trophy fishing (Lych and Remr, 2019). In certain contexts, this can offset or even exceed the economic losses associated with predation on other species.

Wels catfish also has both positive and negative implications for aquaculture. It is farmed commercially in parts of Europe, contributing to aquaculture production and economic activity. It has little impact in aquaculture systems where it is intensively reared in extensive or semi-natural systems. The greatest threat is from potential escape or release from aquaculture into open water bodies and subsequent establishment of wild populations, increasing ecological risks. Additionally, the species can act as a host for pathogens, potentially facilitating disease transmission within aquaculture systems. Disease outbreaks can result in significant economic losses, particularly in intensive farming operations.

Consequently, the overall impact of European catfish is highly context-dependent, requiring careful management to balance ecological integrity with economic interests (Brignone et al., 2026). Its spread has necessitated increased investment in monitoring and management to support implementation of EU Regulation 1143/2014 on invasive alien species – see EU LIFE Predatorfish project (<https://lifepredator.eu/wels-catfish/>). Control measures generally orientate around population

management through capture and removal of individuals, by direct capture methods such as angling and long lining (Vejřík et al., 2019a, b, 2024) or gillnetting and larval trawling (Čech et al., undated), and can be costly and resource-intensive. These costs are particularly relevant in regions where the species is considered invasive and poses a threat to native biodiversity and fisheries.

9.17.3 References

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