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Atlantic salmon and non-native species: is there an issue?

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Introduction

The International Union for the Conservation of Nature (IUCN) assert that the spread of Invasive Alien Species (IAS) is, after habitats loss, the second most significant threat to global biodiversity. The most recent global assessment report on biodiversity and ecosystem services carried out by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) (Díaz *et al.* 2019) concluded that nearly one fifth of the Earth's surface is at risk of plant and animal invasions, impacting native species, ecosystem functions and nature's contributions to people, as well as economies and human health.

Technically, any species which has been transplanted or introduced to an area outwith its native range (even within the same country), can be regarded as being an 'alien species'. However, the terms 'invasive nan-native species' (INNS) or 'invasive alien species' (IAS) are used interchangeably within the scientific literature to describe non-native (or alien) species which have become established and have negatively impacted native biodiversity, as well as ecosystem services on which humans depend. For clarity, the term IAS is used throughout to describe such species.

A number of international IAS databases are available to provide detail on the distribution, ecology and impact of many invasive species. Details of the best known, and most widely accessed, of these are provided in Table 1. Despite the volume of information available, it remains a difficult task to consolidate these data into a single, global, assessment of the number of IAS. Turbelin *et al.* (2017) attempted to map the global state of IAS using existing databases and concluded that, whilst evaluating the extent of such movements at a global level is problematic, it was clear that significant variation exists in the spatial patterns of invasion. This review also indicated that areas contained within the Global North (e.g. more developed countries), along with tropical islands, are the main recipients of IAS. The rise in establishment of IAS both internationally (Turbelin *et al.* 2017) and within Europe (Keller *et al.* 2011) is largely due to increased globalisation, which has led to the increased movement of people and goods to new areas.

Table 1. *Key IAS databases in use within Europe.*

IAS Database name	Access
Global Invasive Species Database (GISD)	http://www.invasivespecies.net/database/
Global Invasive Species Information Network (GISIN)	http://www.gisin.org/
CABI Invasive Species Compendium	https://www.cabi.org/isc/
European Alien Species Information Network (EASIN)	https://easin.jrc.ec.europa.eu/easin
The Regional Euro-Asian Biological Invasions Centre (REABIC)	https://www.reabic.net/
Delivering Alien Invasive Species Inventories for Europe (DAISIE)	http://www.europe-aliens.org/
The European Network on Invasive Alien Species (NOBANIS)	https://www.nobanis.org/

The role of globalisation in IAS spread

In a recent analysis of global introductions Seebens *et al.* (2017) and found that over one third of all introductions in the past 200 years occurred after 1970 and suggests that the rate of introductions is showing no sign of slowing down. Within Europe, Hulme (2009) showed that the highest rates of introductions have occurred over the last 25 years.

The expansion of new species into new areas can happen through natural means, however most invasions are associated with human activity. The pathways for aquatic organisms to enter new waterbodies either within, or into, Europe are many but Keller *et al.* (2017) break these down into three general themes: 1) transportation as a commodity; 2) arrival with a transport vector; and 3) self-dispersal by the species along infrastructure corridors. *Transportation as a commodity* is perhaps the most obvious of the three pathways described in that it includes the movement of fish or other taxa for sport (e.g. stocking and release of bait fish), aquaculture, biological control or aquarium (including pond) use. This also includes the transport of parasites (e.g *Gyrodactylus salaris*) and diseases which are not native to host species.

Arrival with a transport vector includes those species that may arrive as a passenger on transport (such as ships), but are not themselves a commodity. Some of the best examples of this include the transport of species in the ballast water of large transport ships, and include the introduction of Eurasian ruffe (*Gymnocephalus cernuus*), zebra mussel (*Dreissena polymorpha*) and round goby (*Neogobius melanostomus*) to the Great Lakes in 1986, 1988 and 1990 respectively (Pratt, 1988; Hebert *et al.* 1989; Corkum *et al.* 2004).

Self-dispersal of IAS along infrastructure corridors, such as roads and canals or via the movement of water between catchments for hydropower or drinking water is less commonly documented for aquatic species, and fish in particular (but see Panov *et al.* 2009). Perhaps the best known example of fish utilising waterways as an invasion route are the Asian carps (*Hypophthalmichthys nobilis* and *H. molitrix*) which have, since their release from aquaculture ponds in the 1970's, migrated northwards through the navigable waterways within the Mississippi River towards the North American Great Lakes (Ridgway and Bettoli 2017). Within Europe, examples include the movement of invasive fish species through the network of inland waterways within the River Rhine catchment (Leuven *et al.* 2009), and the transport of Arctic charr (*Salvelinus alpinus*) through a 1400m long tunnel used to transport water from Loch Awe to Cruachan Reservoir (Maitland and Adams 2019).

Other routes of invasion exist such as the deliberate release of unwanted plants, fish or invertebrates (such as crayfish), by aquarists and pond keepers who can obtain material from both retailers (West *et al*. 2019) and from a growing number of online suppliers (Lenda *et al.* 2014). Some pathways are less obvious, such as the deliberate release of animals for religious purposes (Liu *et al.* 2012).

Interactions with other stressors

Whilst invasive alien species (IAS) are considered one of the greatest threats to biodiversity, the magnitude of these impacts may be modified through their interactions with other drivers of change (Bellard *et al*. 2016; Didham *et al*. 2005; Early *et al*. 2016; Seebens *et al*. 2017). Other pressures, such as climate change and resource exploitation, may magnify the impact of IAS on native biota, meaning that when considering the impact of IAS one must also consider the role of other environmental stressors which may affect the fitness of native species.

Extreme climatic events resulting from climate change, such as floods can either transport IAS to new areas or allow fish to escape impoundments. This includes put-and-take fisheries or fish farms located in vulnerable floodplain areas. The increased frequency of storm events may also facilitate the escape of farmed fish into the wild, where they may establish new populations or interbreed with wild conspecifics (e.g. Jensen *et al*. 2010).

Climate change can also result in the opening up new pathways of introduction of IAS. For example, reduced shipping times facilitated by new or emerging Arctic shipping passages caused by melting ice caps may increase the risk of IAS surviving the journey, as well as increasing the potential for some species to greatly increase their global range naturally in response to these changing environmental conditions (Nong *et al*. 2018). Aquatic species at high latitudes may experience greater temperature-mediated pressure which may impact their ability to compete with new species, whether or not they have arrived there as a function of range expansion and natural colonisation. Greater environmental tolerances, a key feature of many successful IAS, may also mean that such animals may have the ability to expand more rapidly than native species to higher latitudes and be better adapted to future changes in climate than native species, thereby allowing them to dominate thermally modified habitats.

At a global scale, Arctic warming may also promote the interchange of fish, and other species, between the Pacific and Atlantic oceans (Wisz *et al*. 2015) and this may present new challenges for Atlantic salmon if such species present a predation risk for either themselves or other species on which they depend for food. Within the Atlantic and other marine regions, climate change is also promoting the shifting of marine species in a northerly direction (Perry *et al*. 2005; Lenoir *et al*. 2011), and this may already be affecting the production of Atlantic salmon at sea.

Changes to thermal and hydrological regimes of freshwaters due to climate change are already predicted to affect the distributions and prevalence of salmonids including Atlantic salmon (*Salmo salar*), brown trout (*Salmo trutta*), and Arctic charr (*Salvelinus alpinus*) (Elliott and Elliott 2010; Jonsson and Jonsson 2010; Finstad and Hein 2012; Svenning *et al*. 2016), suggesting that current habitats may be becoming less suitable than previously, and freshwater systems that currently have thermal barriers limiting the establishment of IAS may become more suitable for IAS as the climate changes (Ellender *et al.* 2016).

Impacts on species and ecosystems

Once introduced IAS can have significant direct and indirect impacts on individual species and aquatic ecosystems. In a review of the published literature Gallardo *et al*. (2016) concluded that IAS can reduce the overall abundance of species and, over time, alter the diversity of aquatic communities across a range of functional groups. However, the scale of any impact is likely to be context-dependent and may differ between species and habitats (Ricciardi *et al*. 2013). Invaders can impact individual species through competition for food and space, predation, disease and parasite transfer and genetic introgression. They can also impact ecosystem functioning, by disrupting foodwebs, modifying the physical environment (Ricciardi and MacIsaac 2011; Ricciardi *et al*. 2013; Gallardo *et al*. 2016). Along with the upward trend in the establishment of new species, the projected impacts on biodiversity are also expected to increase because of the growing number of species involved and the increasing vulnerability of ecosystems to invasions resulting from other pressures such as habitat loss, degradation, fragmentation, over-exploitation and climate change (Clavero and García-Berthou 2005; Didham *et al*. 2005; Bellard *et al*. 2016).

Examples of IAS impacts on Atlantic salmon

Little exists within the scientific literature with regard to the direct impact of exotic IAS on Atlantic salmon, however more information is available on the potential impact on ‘Locally Absent Species’ (LAS) (i.e. species which may be native to a nation state, but are not locally native to that waterbody). Short, non-exhaustive, examples of potential relationships between Atlantic salmon and predators, competitors, invertebrates and plants are provided to illustrate a range of interactions which may occur.

Introduced fish - predators

Whilst juvenile Atlantic salmon may have many natural predators, such as piscivorous birds, mammals and fish (Thorstad *et al*. 2012), there is little in the published scientific literature relating to the introduction of new piscivorous predators from outwith Europe. That is not to say, however, that native species, inappropriately introduced, do not present a potential threat. The introduction of pike (*Esox lucius*) to a waterbody from which it is naturally absent may constitute a significant risk to native fish species – including Atlantic salmon. Several studies (e.g. Kennedy *et al*. 2018) have demonstrated the impact that pike may have on Atlantic salmon smolts which migrate through natural lakes. It is logical to assume, therefore, that the introduction of this, or other locally non-native piscivorous fish species (including large trout), to new waterbodies where they may come into contact with Atlantic salmon will result in increased predation pressure.

Introduced fish - competitors

Whilst the introduction of a piscivorous predator may seem an obvious risk to Atlantic salmon (and other native fish species), the introduction and spread of species such as the Eurasian minnow (*Phoxinus phoxinus*) can also pose an unrealised risk. Museth *et al*. (2007), in a study of the impact of the expanding range of Eurasian minnow populations in Norway concluded that this species has the potential to impact brown trout (*Salmo trutta*) populations through increased competition for food resources. Whether this species has an impact on juvenile Atlantic salmon is not known, but given the nature of the competitive interaction with brown trout, potential for some ecological overlap exists. A similar situation exists in Scotland, were this species has been widely, and deliberately, spread by anglers by way of discarded live bait, and as a potential prey source for local trout.

Bullhead *Cottus gobio* are included in Annex II of the EU Habitats Directive and within the UK and they are a conservation feature in 17 Special Areas of Conservation. However, within the UK this species has a restricted, southerly distribution, and it is naturally absent from Scotland. Introduced populations exist in the Clyde, Forth and Tweed catchments and there is concern that competition, both for food and space, between bullheads and juvenile Atlantic salmon may result in an overall negative impact on Atlantic salmon production. Empirical evidence to support this is limited to basic inference, however Gabler *et al*. (2001) in one of the few studies to explore the interaction between Atlantic salmon parr and bullhead reported selective segregation in prey choice between species. It remains possible, however, that higher levels of dietary and spatial overlap exist between Atlantic salmon fry and bullhead at varying stages in their life history.

Introduced invertebrates

Introduced crayfish species can negatively impact salmonid fish (Holm *et al*. 1989; Savino and Miller 1991; Rubin *et al*. 1993; Griffiths *et al*. 2004; Fitzsimons *et al*. 2007; Peay *et al*. 2010). In Europe, most attention has in recent decades focussed on the impact of the signal crayfish *Pacifastacus leniusculus*, a North American species which has been widely introduced and has had significant adverse impacts on freshwater ecosystems and individual species (Holdich *et al*. 2009). Where introduced, crayfish can predate on fish and their eggs, compete for food and shelter and modify instream and bankside habitats (Everard *et al*. 2009; Reynolds 2014; Mathers *et al*. 2016; Turley *et al*. 2017). Whilst there is significant concern over the impact of introduced signal crayfish on native salmonid species, evidence of a significant impact through direct predation of juvenile Atlantic salmon is sparse. Fluvarium-based trials which have investigated whether signal crayfish have the potential to excavate and consume Atlantic salmon redds have been equivocal. One study demonstrated active excavation and predatory loss of eggs and fry (Edmonds *et al*. 2011), one showed no impact (Gladman *et al*. 2012) and another demonstrated that egg predation may only be carried out by larger animals (Findlay *et al*. 2014).

Introduced plants

Invasive riparian plants are now considered to be a significant issue for aquatic ecosystems, and species such as Japanese knotweed *Fallopia japonica* and Himalayan balsam *Impatiens glandulifera* are now widely established on river banks across the northern hemisphere (Seeney *et al*. 2019). Rapidly invading plant species such as these can negatively impact aquatic habitats through increased shading, reduced surface water temperatures and, as a consequence, may result in a reduced potential for autochthonous production. By outcompeting native riparian plant communities, winter dieback of large invasive monocultures can also result in increased flood risk and erosion. Changes to the composition and quantity of allochthonous leaf litter may also affect both the diversity and abundance of aquatic invertebrates (Claeson *et al*. 2013). Clear potential exists for riparian plant IAS to affect the distribution and abundance of Atlantic salmon in affected watercourses, although the scale of any impact will be site-specific and may be dependent on a range of other factors, such as the extent of plant growth, location within individual catchments and stream width.

Interactions between non-native in-stream plants and Atlantic salmon are absent from the peer-reviewed scientific literature, suggesting that this is a low risk. However, where the luxurious growth of riverine plants do become established, they can, in association with low flows, impact adult salmonids (House *et al*. 2016). In Scotland, the establishment of the River water-crowfoot *Ranunculus fluitans*, in the River Spey in 1979 is considered to be a problem for fishery managers. This species, which is native to other parts of the UK but not to the Spey catchment creates problems for anglers by choking pools and making it difficult for anglers to catch and land Atlantic salmon and other salmonids (Laughton *et al*. 2008). The ecological impact of this species on Atlantic salmon, particularly juvenile fish, is unquantified. Elsewhere in Scotland another water crowfoot species (*R. aquatilis*) has been present in the River Dee (Aberdeenshire) for almost 100 years, having first been recorded there in 1916. Where present, similar concerns in relation to plant abundance and impact on angling success exist.

Salmonids as invaders

Salmonids are successful invaders and new populations have been established across the globe over the course of the last century. Brown trout, for instance, is included in the list of the '100 worst invasive alien species' compiled by the IUCN Invasive Species Specialist Group (Lowe *et al*. 2000). Atlantic salmon are themselves, listed as IAS in several of the global IAS databases (IUCN Global Invasive Species database, CABI Invasive Species Compendium, and the NOBANIS European Network on Invasive Alien Species), but these listings relate specifically to the release of domestically reared fish into the wider environment. Farmed Atlantic salmon pose a particular risk to wild individuals because, despite domestication, likely to have similar physiological tolerances, can compete for food and space within rivers, may arrive in large numbers and can spawn with wild conspecifics. The importance of inherited adaptive variation in maintaining Atlantic salmon and other salmonid populations is well understood (Garcia de Leaniz *et al*. 2007; Fraser *et al.* 2011) and a substantial scientific literature exists to demonstrate that the stocking and transfer of Atlantic salmon (and other salmonids) could threaten the maintenance of wild populations (e.g. Karlsson *et al*. 2017). Over time, Bolstad *et al*. (2017) argues that the continued ingress of domesticated genes and the loss of locally adapted traits can alter the life history of wild Atlantic salmon populations. The same process may also affect Atlantic salmon populations which have been supplemented with wild fish from other catchments. Despite these impacts, stocking continues to be utilised as a tool to support declining numbers of returning fish (Aas *et al*. 2018). Instead of the term IAS to describe farmed Atlantic salmon in areas where they can cause damage to wild populations, a new term, Invasive Alien Conspecifics (IAC) could be used to differentiate between the impacts of other species to those caused by the same, but farmed or stocked, individuals. The term IAC could be used to describe the escape or deliberate release of any species into waters where wild conspecifics occur.

Beyond genetic impacts, the transfer of Atlantic salmon, and other salmonids, beyond national boundaries carries the risk of disease and parasite transfer. The introduction of the ectoparasite *G. salaris* to Norwegian rivers from infected Atlantic salmon from Sweden in 1975 (Sandodden *et al*. 2018) remains one of the greatest threats to Atlantic salmon in that country. It remains the best known example of how parasite transfer can have a catastrophic impact on individual Atlantic salmon populations. Whilst the parasite has been successfully removed from several river systems in Norway, the economic costs of this have been high, both in terms of the biological resource lost and the financial cost of eradication and post-treatment monitoring.

Salmonid introductions are often used to demonstrate the impact of biological invasions in aquatic ecosystems because they are among the most widely introduced fish taxa around the globe (Crawford and Muir 2008; Buoro *et al*. 2016). Within their native range salmonid introductions have been primarily driven by the desire to establish or maintain recreational fisheries, although introductions have also been carried out as conservation or compensatory/restoration measures. However the underlying policy instruments used to guide this activity vary between Nation States (Aas *et al*. 2018).

Pacific salmonids, and Rainbow trout *Oncorhynchus mykiss* in particular, have established self-sustaining populations around the globe, and proved to be a highly successful invaders in many cases (Crawford and Muir 2008). In terms of establishing self-sustaining populations in western Europe, the success of Rainbow trout has been limited, with only 130 confirmed or potential self-sustaining populations being recorded across 16 European countries (Stanković *et al*. 2015). Where present the potential for competition with, or predation of, juvenile Atlantic salmon cannot be discounted.

Despite being present in the White Sea and northern Scandinavia since the late 1950's the sudden expansion of odd-year Pink salmon *O. gorbuscha* into rivers in Finland, Norway, Iceland, Scotland, England, Ireland, Denmark, France and Germany in 2017 was unexpected (Armstrong *et al.* 2018; Mo *et al*. 2018; Sandlund *et al*. 2019). Time will tell whether this explosion of Pink salmon records around northwestern Europe was a single event, or whether this will become a more regular occurrence. The large numbers associated with returning Pacific salmonids within their native and introduced range suggest that propagule pressure may be high, but differences in run timing and overall life history may mean that their actual impact on Atlantic salmon is low. However, data to support this is currently lacking.

The Ponto-Caspian problem

IAS originating from the Ponto-Caspian region are considered to be a particular threat to freshwater habitats to which they are not native. The potential impact of species such as Killer shrimp (*Dikerogammarus villosus*), Demon shrimp (*D. haemobaphes*), Zebra mussel, Quagga mussel (*Dreissena r. bugensis*) and round goby on freshwater ecosystems is well documented (e.g. Gallardo and Aldridge 2015). Currently, over 100 species are known to be spreading outwards from the Ponto-Caspian region via a number of invasion pathways (Ketelaars 2004). The impact of Ponto-Caspian species on Atlantic salmon, their habitats and the ecological processes which support them is unquantified, and the potential for introduction remains a significant risk.

The cost of IAS

Approximately 12,000 non-native species have been registered in Europe (Keller *et al.* 2011; European Commission 2014) and many of the 10-15% of those species which have become IAS (Lockwood *et al*. 2013) were introduced intentionally because they have a high value to humans (Ehrenfeld 2010).

Where established, particularly within aquatic habitats, IAS are often difficult or contain, and in many cases, impossible to eradicate. Regardless as to whether an IAS can be controlled or eradicated, the economic cost of dealing with such species can be significant and long-term (Jardine and Sanchirico 2018). Direct management costs, such as those required to implement control measures or, where possible, eradicate can be expensive to implement and may require the allocation of research costs, as well as those required for equipment, monitoring and manpower. Lost benefits through a reduction in production, environmental damage (such as erosion, siltation) and impacts on environmental services (such as water supply or recreational fisheries) can escalate the cost of opportunity losses. The estimated annual costs attributable to the impact of IAS are as high as £1.7 billion in the UK, $1.4 trillion in the USA and $33.5 billion in SE Asia (Pimentel *et al*. 2005; Williams *et al*. 2010). Such evaluations are likely to be under-estimates of the true cost. In fact, Pimentel *et al.* (2001) suggests that losses in global production due to the impact of IAS could be as high as 5%.

Notwithstanding the fact that IAS can carry significant economic costs, biological invasions represent one of the greatest threats to freshwater ecosystems (Ehrenfeld 2010), and the ecosystem services that they provide (Pejchar and Mooney 2009).

Legislation and policy

International agreements

The IPBES Global Assessment Report on Biodiversity and Ecosystem Services published in May 2019 (Díaz *et al*. 2019) reaffirmed the role that IAS play in species extinctions worldwide. Whilst the scale of impact is of growing concern, the dangers posed by IAS at a global level have been recognised for some time. Article 8(h) of the Convention on Biological Diversity (CBD) states that, *'Each contracting Party shall, as far as possible and as appropriate, prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species'*, and as long ago as 1992, the CBD put forward a three-stage hierarchical approach to deal with the issue (Prevention, Early detection and rapid eradication; and Control and/or containment). In 2010 the majority of the world's governments adopted the CBD Strategic Plan for Biodiversity. This included the 20 headline objectives which now commonly referred to as the referred the 'Aichi Targets'. One of these, Aichi Target 9, is specifically related to the control of IAS and states that: *'By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated and measures are in place to manage pathways to prevent their introduction and establishment'*. This international commitment to addressing IAS was re-asserted in 2015 through the CBD's 2030 Agenda for Sustainable Development, which states that: *'By 2020, introduce measures to prevent the introduction and significantly reduce the impact of invasive alien species on land and water ecosystems and control or eradicate the priority species'*.

In response to global concerns, and driven by the fact that the number and extent of IAS were increasing within Europe, the European Union published two regulations of relevance to the issue of IAS and Atlantic salmon. The first of these was the Council Regulation (EC) No 708/2007 concerning use of alien and locally absent species in aquaculture. This regulation is intended to provide a framework to ensure adequate protection of aquatic habitats from the risks associated with the use of non-native species in aquaculture. The second is EU Regulation 1143/2014 on invasive alien (non-native) species, this is a commitment contained in Target 5 of the EU's Biodiversity Strategy to 2020 (European Commission, 2011). The Regulation establishes a coordinated EU-wide framework for action to prevent, minimise and mitigate the adverse impacts of IAS on biodiversity and ecosystem services, and limit their damage to the economy and human health. The Regulation also lists 49 species of Union Concern, many of which are aquatic. Examples include Chinese mittencrab *Eriocheir sinensis*, Spiny-cheek crayfish *Orconectes limosus*, Virile crayfish *O. virilis*, Signal crayfish, Red swamp crayfish *Procambarus clarkii*, Marbled crayfish P. *fallax f. virginalis* and Amur sleeper *Percottus glenii*. Riparian plant species, such as Himalayan balsam *Impatiens glandulifera* and Giant hogweed *Heracleum mantegazzianum* as well as a number of species which are commonly associated with large river and standing water habitats.

These new regulations augment existing commitments to control IAS within other international agreements (e.g. Bonn Convention on the Conservation of Migratory Species of Wild Animals and Bern Convention on the Conservation of European Wildlife and Natural Habitats), as well as existing European instruments for nature conservation (Article 22 of the EC Habitats Directive and Article 11 of the Birds Directive 79/409/EC). Whilst not explicitly mentioned in legislation, the EC Water Framework Directive requires member States to put in place national measures to achieve or maintain a good ecological status for European inland, transitional and coastal waters by 2015 and prevent their further deterioration. The control or eradication of IAS forms part of this work.

National measures

National controls on the spread on IAS at a national level vary significantly throughout Europe and, in the case of the UK, measures vary between national administrations. McGeoch *et al*. (2010) calculated that 55% of CBD signatories had domestic legislative measures in place to deal with IAS and many European countries have national regulations in place to regulate the import of species (e.g. McGeogh *et al*. 2016). However, the EU

IAS Regulation presents a new framework for coordinated action across all Member States, with prevention, eradication and control measures being applied at the national level, along with a commitment to identify invasion pathways and deal with new invaders rapidly. Information sharing between Member States and new reporting obligations should lead to greater consistency of approach when dealing with IAS in future.

An important, but less well appreciated, aspect of IAS introductions is the 'translocation' (including stocking) of species within a nation state, i.e. from waters in which a species occurs naturally to waters where it does not occur naturally (Bean and Copp 2016; Dodd *et al*. 2019). A long history of unregulated and unrecorded fish movements over past centuries makes it difficult to identify the true native range of some species. The deliberate introduction of 'locally absent' species may pose an equally high risk to native species and ecosystems as the introduction of alien species from other countries, and this is recognised internationally (ICES 2004; EIFAAC 2007). Three examples are provided in Section 5, namely pike, Eurasian minnow and bullhead.

Conclusion

Species which live in northerly latitudes are under significant threat from both climate change and IAS (Walther *et al*. 2009). The climate is warming faster at northern latitudes than elsewhere suggesting that the added pressure of IAS may be particularly significant in the temperate and Arctic waters which support vulnerable populations of Atlantic salmon.

The success or failure of IAS establishment depends on a number of factors, including the ecological tolerance of the invader and the environmental resistance of invaded systems (Ricciardi and Atkinson 2004), the frequency and intensity of propagule pressure (Lockwood *et al*. 2005), and the plasticity of invaders (Westley 2011). Although much more is known about the ecological consequences or impacts of biological invasions than in the past, such knowledge is only gained after AIS have become well established (Dunham *et al*. 2002). This points towards a need for greater surveillance, by Government, fishery managers and the general public to identify new arrivals as soon as possible. Once established, the ecological impacts of IAS are difficult to reverse (Vitule *et al*. 2009), particularly in freshwater habitats where eradication and control measures which do not, in themselves, cause significant environmental damage, are relatively few.

Introduced Atlantic salmon, either from fish farm escapes or planned stocking can have significant impacts of wild populations. These AIC (Aquatic Invasive Conspecifics), can pose a particular risk, and if the Atlantic salmon aquaculture industry moves northward in response to climate change, there is potential to affect populations which are currently not exposed to significant pressure.

Looking forward – horizon scanning and Risk Assessment

Horizon scanning is an approach used to prioritise the threat posed by potentially new IAS not yet established within a region, and has been seen as an essential component of IAS management with demonstrated net economic and ecological benefits (Roy *et al*. 2019). When used alongside risk analysis (Roy *et al*. 2017; Dodd *et al*. 2018), the impact of potential IAS on Atlantic salmon can be evaluated in a systematic and consistent way between nation states. The scientific literature suggests that risk analysis for Atlantic salmon across its native range would be a significant step forward in predicting the future impact of IAS on this species. National risk assessments exist for many of the most damaging and widely distributed IAS, as well as contingency plans to deal with parasites and notifiable diseases which may affect Atlantic salmon rivers. It is important that horizon scanning does not limit its focus to species which are 'exotic' to a locality, but also includes an assessment of the potential for Locally Absent Species (LAS) to impact Atlantic salmon directly, or the habitats and ecological processed that support them. Experience has shown, in the case of *G. salaris*, that 'local species' can act as reservoirs of disease and parasites and that their movement between catchments can pose a significant risk.

Monitoring and surveillance

A number of international agreements at both a global and European level provide clear direction with regard to the importance of the IAS issue and our collective responsibility to act. The EU Regulation 1143/2014 on invasive alien (non-native) species, requires greater surveillance for IAS and the identification of pathways of invasion. Whilst the cost of surveillance can be high, the adoption of new tools, such as eDNA for monitoring invasive fish (e.g. Gustavson *et al*. 2015), invertebrates (e.g. Harper *et al*. 2018) and even *G. salaris* (Rusch *et al*. 2018) can both reduce the costs and ease of sampling.

In summary, IAS, along with other stressors can have the potential to have a negative impact on Atlantic salmon, although direct empirical evidence is lacking in many areas. Whilst legislation to control the further movement and surveillance on IAS has improved significantly within Europe, promising better coordination between Member States, there remains a need to develop a strategy for Atlantic salmon, which sets out the risks posed by IAS, AIC and Locally Absent Species against a range of climate change scenarios and known pathways for invasion. The lack of published empirical evidence on the actual impact of AIS on Atlantic salmon remains a cause of concern, and any strategy should consider how this may be addressed in future years.

References

Aas, Ø., Cucherousset, J., Fleming, I.A., Wolter, C., Höjesjö, J., Buoro, M., Santoul, F. *et al*. 2018. Salmonid stocking in five North Atlantic jurisdictions: Identifying drivers and barriers to policy change. Aquatic Conservation: Marine Freshwater Ecosystems, 28: 1451– 1464.

Armstrong, J.D., Bean, C.W. and Wells, A. 2018. The Scottish invasion of pink salmon in 2017. Journal of Fish Biology, 93: 8–11.

Bean, C.W. and Copp, G.H. 2016. Aquatic Alien Species and the WFD: Proposed list of 'Locally Non-Native' Species and Guidance on its Interpretation. WFD UK Technical Advisory Group. http://www.wfduk.org/sites/default/files/Media/Environmental%20standards/Locally%20non-native%20species%20-%20revised%20September%202016.pdf

Bellard, C., Cassey, P. and Blackburn, T. M. 2016. Alien species as a driver of recent extinctions. Biology Letters, 12: 20150623.

Bolstad, G.H. Hindar, K., Robertsen, G., Jonsson, B., Sægrov, H., Diserud, O.H.,Fiske, P., Jensen, A.J. *et al*. 2017. Gene flow from domesticated escapes alters the life history of wild Atlantic salmon. Nature Ecology and Evolution, 1: 0124.

Buoro, M., Olden, J.D. and Cucherousset, J. 2016. Global Salmonidae introductions reveal stronger ecological effects of changing intraspecific compared to interspecific diversity. Ecology Letters, 19: 1363–1371.

Claeson, S.M., LeRoy, C.J., Barry, J.R. and Kuehn, K.A. 2013. Impacts of invasive riparian knotweed on litter decomposition, aquatic fungi, and macroinvertebrates. Biological Invasions, 16: 1531–1544.

Clavero, M. and García-Berthou, E, 2005. Invasive species are a leading cause of animal extinctions. Trends in Ecology and Evolution, 20: 110-.

Corkum, L.D., Sapota, M.R. and Skora, K.E. 2004. The Round Goby, *Neogobius melanostomus*, a fish invader on both sides of the Atlantic Ocean. Biological Invasions, 6: 173-181.

Crawford, S.S. and Muir, A.M. 2007. Global introductions of salmon and trout in the genus *Oncorhynchus*: 1870–2007. Reviews in Fish Biology and Fisheries, 18: 313–344.

Díaz, S., Josef Settele, J., Brondízio, E., Ngo, H.T., Guèze, M., Agard, J., Arneth, A., *et al*. 2019. Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://www.ipbes.net/sites/default/files/downloads/spm_unedited_advance_for_posting_htn.pdf

Didham, R.K., Tylianakis, J.M. Hutchison, M.A. Ewers, R.M. and Gemmell, N.J. 2005. Are invasive species the drivers of ecological change? Trends in Ecology and Evolution, 20: 470-474.

Dodd, J., Vilizzi, L., Bean, C.W., Davison, P.I. and Copp, G.H. 2019. At what spatial scale should risk screenings of translocated freshwater fishes be undertaken – river basin district or climo-geographic designation? Biological Conservation, 230: 122-130.

Early, R., Bradley, B.A., Dukes, J.S., Lawler, J.J., Olden, J.D., Blumenthal, D.M., Gonzalez, P., *et al*. 2016. Nature Communications, 2016: 7: 12485.

Edmonds, N.J., Riley W.D. and Maxwell D.L. 2011. Predation by *Pacifastacus leniusculus* on the intra-gravel embryos and emerging fry of *Salmo salar*. Fisheries Management and Ecology, 18: 521–524.

Ehrenfeld, J.G. 2010. Ecosystem consequences of biological invasions. Annual Review of Ecology, Evolution, and Systematics, 41: 59-80.

EIFAC. 2007. EIFAC Ad hoc Working Party on Introductions and Stocking. Available at: ftp://ftp.fao.org/fi/body/eifac/eifac.asp

Ellender, B.R., Rivers-Moore, N.A., Coppinger, C.R., Bellingan, T.A. and Weyl, O.L.F. 2016. Towards using thermal stress thresholds to predict salmonid invasion potential. Biological Invasions, 18: 513–525.

Elliott, J.M. and Elliott, J.A. 2010. Temperature requirements of Atlantic salmon *Salmo salar*, brown trout *Salmo trutta* and Arctic char *Salvelinus alpinus*: predicting the effects of climate change. Journal of Fish Biology, 77: 1793–1817.

European Commission. 2011. The EU Biodiversity Strategy to 2020. Available at: http://ec.europa.eu/environment/nature/info/pubs/docs/brochures/2020%20Biod%20brochure%20final%20lowres.pdf

European Commission. 2014. Invasive Alien Species: A European response. Available at: http://ec.europa.eu/environment/nature/invasivealien/docs/ias-brochure-en-web.pdf

Everard M., Gray J., Wilkins-Kindemba V. and Cowx I.G. 2009. Impacts of invasive species on ecosystem services: the case of the signal crayfish (*Pacifastacus leniusculus*). Environmental Law Management, 21: 250–259.

Fausch, K.D. 2007. Introduction, establishment and effects of non-native salmonids: considering the risk of rainbow trout invasion in the United Kingdom. Journal of Fish Biology, 71: 1–32.

Findlay, J.D.S., Riley, W.D. and Lucas, M.C. 2015. Signal crayfish (*Pacifastacus leniusculus*) predation upon Atlantic salmon (*Salmo salar*) eggs. Aquatic Conservation: Marine Freshwater Ecosystems, 25: 250– 258.

Finstad, A.G. and Hein, C.L. 2012. Migrate or stay: terrestrial primary productivity and climate drive anadromy in Arctic char. Global Change Biology, 18: 2487–2497.

Fitzsimons, J.D., Jonas, J.L., Claramunt, R.M., Williston, B., Williston, G., Marsden, J.E., Ellrott, B.J. and Honeyfield, D.C. 2007. Influence of egg predation and physical disturbance on lake trout *Salvelinus namaycush* egg mortality and implications for life-history theory. Journal of Fish Biology, 71: 1–16.

Fraser, D.J., Weir, L.K., Bernatchez, L., Hansen, M.M., and Taylor, E.B. 2011. Extent and scale of local adaptation in salmonid fishes: Review and meta-analysis. Heredity, 106: 404–420.

Gabler, H.M., Amundsen, P.A. and Herfindal, T. 2001. Diet segregation between introduced bullhead (*Cottus gobio* L.) and Atlantic salmon parr (*Salmo salar* L.) in a sub-Arctic river. Archiv Fur Hydrobiologie, 151: 609-625.

Gallardo, B. and Aldridge, D.C. 2015. Is Great Britain heading for a Ponto–Caspian invasional meltdown? Journal of Applied Ecology, 52: 41-49.

Gallardo, B., Clavero, M., Sánchez, M.I. and Vilà, M. 2016. Global ecological impacts of invasive species in aquatic ecosystems. Global Change Biology, 22: 151-163.

Garcia de Leaniz, C., Fleming, I.A., Einum, S., Verspoor, E., Jordan, W.C., Consuegra, S. and Quinn, T.P. 2007. A critical review of inherited adaptive variation in Atlantic salmon. Biological Reviews, 82: 173–211.

Gladman, Z.F., Adams, C.E., Bean, C.W., Long, J. and Yeomans, W.E. 2012. Investigating the threat of non-native North American signal crayfish (*Pacifastacus leniusculus*) to salmon redds. Aquatic Conservation: Marine and Freshwater *Ecosystems*, 22: 134–137.

Griffiths S.W., Collen P. and Armstrong J.D. 2004. Competition for shelter among over-wintering signal crayfish and juvenile Atlantic salmon. Journal of Fish Biology, 65: 436–447.

Gustavson, M.S., Collins, P.C., Finarelli, J.A., Egan, D., Conchúir, R.Ó., Wightman, G.D., King, J.J. *et al.* 2015. An eDNA assay for Irish *Petromyzon marinus* and *Salmo trutta* and field validation in running water. Journal of Fish Biology, 87: 1254-1262.

Harper, K.J., Anucha, P.N., Turnbull, J.F., Bean, C.W. and Leaver, M.J. 2018 Searching for a signal: Environmental DNA (eDNA) for the detection of invasive signal crayfish, *Pacifasticus leniusculus* (Dana, 1852). Management of Biological Invasions, 9: 137-148.

Hebert, P.D.N., Muncaster, B.W. and Mackie, G.L. 1989. Ecological and genetic studies on *Dreissena polymorpha* (Pallas) - a new mollusk in the Great Lakes. Canadian Journal of Fisheries and Aquatic Sciences, 46: 1587–1591.

Holdich D.M., Reynolds J.D., Souty-Grosset C. and Sibley P.J. 2009. A review of the ever increasing threat to European crayfish from non-indigenous crayfish species. Knowledge and Management of Aquatic Ecosystems, 11: 394–395.

Holm, J.C. 1989. Atlantic salmon parr (*Salmo salar*) and noble crayfish (*Astacus astacus*) in duoculture. Aquacultural Engineering, 8: 79–94.

House, A.R., Thompson, J.R., Roberts, C., Smeth, K., Old, G. and Acreman, M.C. 2017. Projecting impacts of climate change on habitat availability in a macrophyte dominated Chalk River. Ecohydrology, 10:e1823.

Hulme, P.E. 2009. Trade, transport and trouble: Managing invasive species pathways in an era of globalization. Journal of Applied Ecology, 46: 10–18.

ICES. 2004. ICES Code of Practice on the Introductions and Transfers of Marine Organisms 2003. International Council for the Exploitation of the Seas, Copenhagen.

Jardine, S.L. and Sanchirico, J.N. 2018. Estimating the cost of species control. Journal of Environmental Economics and Management, 87: 242-257.

Jensen, O., Dempster, T., Thorstad, E.B., Uglem, I. and Fredheim, A. 2010. Escapes of fishes from Norwegian sea-cage aquaculture: causes, consequences and prevention. Aquaculture Environment Interactions, 1: 71-83.

Jonsson, B. and Jonsson, N. 2009. A review of the likely effects of climate change on anadromous Atlantic salmon *Salmo salar* and brown trout *Salmo trutta*, with particular reference to water temperature and flow. Journal of Fish Biology, 75: 2381-2447.

Karlsson, S., Diserud, O.H., Fiske, P. and Hindar, K. 2016. Widespread genetic introgression of escaped farmed Atlantic salmon in wild salmon populations. ICES Journal of Marine Science, 73: 2488–2498.

Keller, R.P., Geist, J., Jeschke, J.M and Kühn, I. 2011. Invasive species in Europe: ecology, status, and policy. Environmental Sciences Europe, 23: 23-39.

Kennedy, R.J., Rosell, R., Millane, M., Doherty, D. and Allen, M. 2018. Migration and survival of Atlantic salmon *Salmo salar* smolts in a large natural lake. Journal of Fish Biology, 93: 134– 137.

Ketelaars, H.M. 2004. Range extensions of Ponto-Caspian aquatic invertebrates in Continental Europe. In: (H. Dumont, T. Shiganova and U. Niermann (Eds.)) Aquatic Invasions in the Black, Caspian, and Mediterranean Seas. pp. 209–236. Springer, The Netherlands.

Laughton, R., Cosgrove, P., Hastie, L. and Sime, I. 2008. Effects of aquatic weed removal on freshwater pearl mussels and juvenile salmonids in the River Spey, Scotland. Aquatic Conservation: Marine Freshwater Ecosystems, 18: 44-54.

Lenda, M., Skórka, P., Knops, J.M.H., Morón, D., Sutherland, W.J., Kuszewska, K. and Woyciechowski, M. 2014. Effect of the internet commerce on dispersal modes of invasive alien species. *PLoS One*, 9 p. e99786.

Lenoir, S., Beaugrand, G. and Lecuyer, E. 2011. Modelled spatial distribution of marine fish and projected modifications in the North Atlantic Ocean. Global Change Biology, 17: 115-129.

Leuven, R.S.E.W., van der Velde, G., Snijders, J., van der Zwart, C., Lenders, R.H.J. and bij de Vaate, A. 2009. The river Rhine: a global highway for dispersal of aquatic invasive species. Biological Invasions, 11: 1989-2008.

Liu, X., McGarrity, M.E. and Li, Y. 2012. The influence of traditional Buddhist wildlife release on biological invasions. Conservation Letters, 5: 107-114.

Lockwood, J.L., Hoopes, M.F. and Marchetti, M.P. 2013. Invasion Ecology. Wiley, Chichester, 428pp.

Lowe, S., Browne, M., Boudjelas, S. and De Poorter, M. 2000. 100 of the world's worst invasive alien species: a selection from the global invasive species database. Published by The Invasive Species Specialist Group (ISSG) a specialist group of the Species Survival Commission (SSC) of the World Conservation Union (IUCN). Available at: www.issg.org/booklet.pdf.

Maitland P.S. and Adams, C.E. 2019. Arctic charr in the Lochs of Scotland: An Assessment of Distribution and Status. Fast-Print Publishing, Peterborough. 382pp.

Mathers, K.L., Chadd, R.P., Dunbar, M.J., Extence, C.A., Reeds, J., Rice, S.P. and Wood, P.J. 2016. The long-term effects of invasive signal crayfish (*Pacifastacus leniusculus*) on instream macroinvertebrate communities. Science of the Total Environment, 556: 207-218.

McGeoch, M.A., Butchart, S.H.M., Spear, D., Marais, E., Kleynhans, E.J., Symes, A., Chanson, J. and Hoffmann, M. 2010. Global indicators of biological invasion: species numbers, biodiversity impact and policy responses. Diversity and Distributions, 16: 95–108.

McGeoch, M.A., Genovesi, P., Bellingham, P.J., Costello, M.J., McGrannachan, C. and Sheppard, A. 2016. Prioritizing species, pathways, and sites to achieve conservation targets for biological invasion. Biological Invasions, 18: 299–314.

Mo, T.A., Thorstad, E.B., Sandlund, O.T., Berntsen, H.H., Fiske, P. and Uglem, I. 2018. The pink salmon invasion: a Norwegian perspective. Journal of Fish Biology, 93: 5– 7.

Moyle, P.B., Kiernan, J.D., Crain, P.K. and Quinones, R.M. 2013. Climate change vulnerability of native and alien freshwater fishes of California: a systematic assessment approach. *PLoS ONE*, 8:e63883.

Museth, J., Hesthagen, T., Sandlund, O.T., Thorstad, E.B. and Ugedal, O. 2007. The history of the minnow *Phoxinus phoxinus* (L.) in Norway: from harmless species to pest. Journal of Fish Biology, 71: 184-195.

Nong, D., Countryman, A.M., Warziniack, T. and Grey, E.K. 2018. Arctic Sea Routes: Potential New Pathways for Nonindigenous Species Spread. Arctic, 71: 269-280.

Panov, V.E., Alexandrov, B., Arbaciauskas, K., Rosa Binimelis, R., Copp, G.H., Grabowski, M., Frances Lucy, F., *et al*. 2009. Assessing the risks of aquatic species invasions via European inland waterways: from concepts to environmental indicators. Integrated Environmental Assessment and Management, 5: 110–126.

Peay S., Guthrie N., Spees J., Nilsson E. and Bradley P., 2009. The impact of signal crayfish (*Pacifastacus leniusculus*) on the recruitment of salmonid fish in a headwater stream in Yorkshire, England. Knowledge and Management of Aquatic Ecosystems, 12: 394-395.

Pejchar, L. and Mooney, H.A. 2009. Invasive species, ecosystem services and human well-being. Trends in Ecology and Evolution, 24: 497-504.

Perry, A.L., Low, P.J., Ellis, J. R. and Reynolds, J.D. 2005. Climate change and distribution shifts in marine fishes. *Science*, 308: 1912-1915.

Pimentel, D., Zuniga, R. and Morrison, D. (2005). Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics, 52: 273–288.

Pimentel, D., McNair, S., Janecka, J., Wightman, J., Simmonds, C., O'Connell, C., Wong, E. *et al*. 2001. Economic and environmental threats of alien plant, animal, and microbe invasions. Agriculture, Ecosystems and Environment, 84: 1–20.

Pratt, D. 1988. Distribution and population status of the ruffe (*Gymnocephalus cernua*) in the St. Louis estuary and Lake Superior. Great Lakes Fishery Commission Research Completion Report. Wisconsin Department of Natural Resources, Superior, WI, 11 pp.

Reynolds, J.D. 2014. A review of ecological interactions between crayfish and fish, indigenous and introduced. Knowledge and Management of Aquatic Ecosystems, 401: 2011 10.

Ricciardi, A. and Atkinson, S.K. 2004. Distinctiveness magnifies the impact of biological invaders in aquatic ecosystems. Ecology Letters, 7: 781–981.

Ricciardi, A. and MacIsaac, H.J. 2011. Impacts of biological invasions on freshwater ecosystems. In: D.M. Richardson. (Ed) Fifty Years of Invasion Ecology. Wiley-Blackwell, Chichester, pp. 211–224.

Ricciardi, A., Hoopes, M.F., Marchetti, M.P. and Lockwood, J.L. 2013. Progress toward understanding the ecological impacts of non native species. Ecological Monographs, 83: 263–282.

Ridgway, J.L. and Bettoli, P.W. 2017. Distribution, Age Structure, and Growth of Bigheaded Carps in the Lower Tennessee and Cumberland Rivers. Southeastern Naturalist, 16: 426-442.

Roy, H.E., Peyton, J., Aldridge, D.C., Bantock, T., Blackburn, T.M., Britton, R., Clark, P. *et al*. 2014, Horizon scanning for invasive alien species with the potential to threaten biodiversity in Great Britain. Global Change Biology, 20: 3859-3871.

Roy, H.E., Rabitsch, W., Scalera, R., Alan Stewart, A., Gallardo, B., Genovesi, P., Essl, F. *et al*. 2018. Developing a framework of minimum standards for the risk assessment of alien species. Journal of Applied Ecology, 55: 526– 538.

Rubin J.F. and Svensson M. 1993. Predation by the Noble crayfish, *Astacus astacus* (L.), on emerging fry of Sea Trout, *Salmo trutta* (L.). Nordic Journal of Freshwater Research, 68: 100–104.

Rusch, J.C., Hansen, H., Strand, D.A., Markussen, T., Hytterød, S. and Vrålstad, T. 2018. Catching the fish with the worm: a case study on eDNA detection of the monogenean parasite *Gyrodactylus salaris* and two of its hosts, Atlantic salmon (*Salmo salar*) and rainbow trout (*Oncorhynchus mykiss*). Parasites and Vectors, 11: 333-345.

Sandlund, O.T., Berntsen, H.H., Fiske, P., Kuusela, J., Muladal, R., Niemela, E., Uglem, I. *et al.* Pink salmon in Norway: the reluctant invader. Biological Invasions, 21: 1033–1054.

Sandodden, R., Brazier, M., Sandvik, M., Moen, A. Wist, A.N. and Adolfsen, P. 2018. Eradication of G*yrodactylus salaris* infested Atlantic salmon (*Salmo salar*) in the Rauma River, Norway, using rotenone. Management of Biological Invasions, 9: 67–77.

Savino J.F. and Miller J.E. 1991. Crayfish (*Orconectes virilis*) feeding on young Lake Trout (*Salvelinus namaycush*): effect of rock size. Journal of Freshwater Ecology, 6: 161–170.

Seebens, H., Blackburn, T.M., Dyer, E.E., Genovesi, P., Hulme, P.E., Jeschke, J.M., Pagad, S. *et al*. 2017. No saturation in the accumulation of alien species worldwide. Nature Communications, 8: 14435.

Seeney, A., Pattison, Z., Willby, N.J., Boon, P.J. and Bull, C.D. 2019. Stream invertebrate diversity reduces with invasion of river banks by non-native plants. Freshwater Biology, 64: 485– 496.

Shine, C., Kettunen, M., Genovesi, P., Gollasch, S., Pagad, S. and Starfinger, U. 2008. Technical support to EU strategy on invasive species (IAS) – Policy options to control the negative impacts of IAS on biodiversity in Europe and the EU. Final report for the European Commission. Institute for European Environmental Policy (IEEP), Brussels, Belgium. 104 pp.

Stanković, D. Crivelli, A.J. and Snoj, A. 2015. Rainbow Trout in Europe: Introduction, Naturalization, and Impacts. Reviews in Fisheries Science and Aquaculture, 23: 39-71.

Svenning, M.A., Sandem, K., Halvorsen, M., Kanstad-Hansen, Ø., Morten Falkegård, M. and Borgstrøm, R. 2016. Change in relative abundance of Atlantic salmon and Arctic charr in Veidnes River, Northern Norway: a possible effect of climate change? Hydrobiologia, 783: 145-158.

Thorstad, E.B., Whoriskey, F., Uglem, I., Moore, A., Rikardsen, A.H. and Finstad, B. 2012. A critical life stage of the Atlantic salmon *Salmo salar*: behaviour and survival during the smolt and initial post-smolt migration. Journal of Fish Biology, 81: 500-542.

Turbelin, A.J, Malamud, B.D. and Francis, R.A. 2017. Mapping the global state of invasive alien species: patterns of invasion and policy responses. Global Ecology and Biogeography, 26: 78–92.

Turley, M.D., Bilotta, G.S., Gasparrini, A., Sera, F., Mathers, K.L., Humpheryes, I. and England, J. 2017. The effects of non-native signal crayfish (*Pacifastacus leniusculus*) on fine sediment and sediment-biomonitoring. Science of the Total Environment, 601-602: 186-193.

Vitule, J.R., Freire, C.A. and Simberloff, D. 2009. Introduction of non-native freshwater fish can certainly be bad. Fish and Fisheries, 10: 98-108.

Walther GR, Roques A, Hulme PE, Sykes MT, Pyšek P, Kühn I, Zobel M. *et al*. 2009. Alien species in a warmer world: risks and opportunities. Trends in Ecology and Evolution, 24: 686–693.

West, A.M., Jarnevich, C.S., Young, N.E. and Fuller, P.L. 2019. Evaluating Potential Distribution of High-Risk Aquatic Invasive Species in the Water Garden and Aquarium Trade at a Global Scale Based on Current Established Populations. Risk Analysis, 39: 1169-1191.

Westley, P.A.H. 2011. What Invasive Species Reveal about the Rate and Form of Contemporary Phenotypic Change in Nature. The American Naturalist, 177: 496-509.

Williams, F., Eschen, R., Harris, D., Djeddour, C., Pratt, R.S., Varia, S., Lamontagne-Godwin, J., Thomas, S.E. and Murphy, S.T. 2010. The Economic Cost of Invasive Non-Native Species on Great Britain. https://secure.fera.defra.gov.uk/nonnativespecies/downloadDocument.cfm?id=487

Wisz, M.S., Broennimann, O., Grønkjær, P., Møller, P.R., Olsen, S.M., Swingedouw, D., Hedeholm, R.B., *et al.* 2015. Arctic warming will promote Atlantic–Pacific fish interchange. *Nature Climate Change*, 5: 261–265.