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T4.2.2 Role of barriers in managing aquatic invasive species

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Executive summary

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Modification of river systems by humans, with the purpose of, for example, irrigation, electricity generation, navigation or flood control, severely fragments freshwater habitats and is widely acknowledged to affect the ecological integrity of river systems. However, with an increasing frequency of river system invasions by non-native species, comes the need to control the spread of Aquatic Invasive Species (AIS). Construction of barriers to intentionally fragment river habitats, or the decision to not remove existing ones, may be needed to control AIS when eradication is unfeasible and when harmful, non-native species should be prevented passage over the barrier, while locally occurring biota should not be hindered in their free movement. Therefore, this represents an increasingly significant aspect in the adaptive management of barriers in rivers and should form part of the decision process for removing barriers, building new ones, or installing biota passes.

Increasingly, the concepts of ‘intentional fragmentation’, ‘management by isolation’ and ‘selective passage’ are being adopted in river catchments threatened by the spread of AIS. In rivers, plants and



microorganisms generally spread passively with the flow or are carried by mobile organisms, whereas some animals are capable of upstream spread, control of which may be possible in some cases using barriers. Selective passage is based on the concept of ecological filters which act on biological traits similar to different parts of a niche space. In this report, a global review is presented of how barriers of different types (for example, physical, physiological, behavioural) are used to control the spread of AIS, of which, physical barriers are most pertinent to the AMBER project. In an evidence-based approach, barriers acting on a range of biological traits are discussed for their effectiveness in limiting the expansion of invasives in the freshwater environment, while still allowing free movement by native biota. Based on a scoring system used for invasive species' impact in Europe, invasive freshwater biota for each major taxonomic group are addressed and recommendations are given on how to manage those respective biota adequately using barriers. This report aims to contribute to raise awareness for exotic freshwater species with freshwater managers, and to help understand how to control the spread of such invasive animals while keeping wider ecosystem impacts to a minimum.

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1 IMPORTANCE OF CONNECTIVITY FOR BIOTA

Increased landscape connectivity has widely been recognized as crucial for the survival of a wide range of terrestrial organisms, which are dependent on food availability, habitat quality and configuration of suitable, connected habitat within the landscape for free movement (Galpern *et al.* 2011; Minor and Gardner 2011; Haddad *et al.* 2015). The ability to move freely across landscapes is required for locating foraging and resting grounds, and for seeking shelter against predation or adverse environmental conditions. It is also crucial for periodic migrations for reproduction (Brodie *et al.* 2015). With increasing anthropogenic influences to the natural environment (for example, fences, road networks and urban developments), habitats are becoming smaller and more isolated and as a result, species are more likely to decline (Didham 2017; Tucker *et al.* 2018). Increasing the connectivity, size and number of habitat patches or networks of protected areas may provide refuges for species moving across the landscape, and corridors for much-needed dispersal opportunities between protected areas (DeFries *et al.* 2007; Hole *et al.* 2009; Baxter-Gilbert *et al.* 2015). Preservation of these movement pathways to modify habitat isolation in order to mitigate negative effects of fragmentation and restore lost habitat, has been widely attempted by creation of wildlife corridors (Haddad *et al.* 2015).

Considering aquatic environments, the importance of connectivity is especially high given the linearity within the (dendritic) profile of river systems. In intensely developed rivers, hydrologic pathways may become fragmented by construction of weirs, dams and water diversion structures, while animals depend on free movement within aquatic systems (longitudinally, but potentially also laterally to floodplains) for foraging, seeking shelter, reproduction and nursery areas (Lucas and Baras 2001). Equally, dispersal is a crucial process for all organisms, including plants and microorganisms. By reconnecting fragmented habitats, biodiversity is often enhanced due to metapopulation processes occurring more frequently, and through restoration of gene flow among populations (Fullerton *et al.* 2010) and by reinstating natural river processes that support habitat diversity downstream of former river obstacles. Compared to movement in the terrestrial environment, or even in large open, slow-moving aquatic environments (marine, lacustrine), the movement of many river organisms is normally constrained to within the river channel and floodplain. Upstream movement of many biota except large free-swimming organisms such as fish, is quite restricted (Kappes and Haase 2012). Even for most fish, their ability to overcome in-river obstacles such as dams, weirs, culverts is often relatively poor, especially in an upstream direction (Hall *et al.* 2011). Downstream movement of plants and animals past barriers is less affected in general, but can still be determined by the hydrological conditions and by behavioural responses to animals to such structures (Zhang *et al.* 2007). The movements of some groups, such as aquatic insects with a winged imago stage, are much less affected by physical barriers (Tonkin *et al.* 2014; Sondermann *et al.* 2015). To enhance hydrologic connectivity in river restoration (for example, in the REFORM project - <https://reformrivers.eu/>), a variety of mechanisms have been adopted widely, including the removal of infrastructure, reducing the impeding effect of in-river structures on free movement by installing various types of fishways and other biota passage corridors (Verástegui *et al.* 2017; Silva *et al.* 2018) and changing reservoir management so that more natural flow regimes are adopted (Schmidt and Wilcock 2008).

Benefits on populations of aquatic wildlife of reducing habitat fragmentation by riverine infrastructure, or even complete removal of such anthropogenic features, have been well documented (for example, Birnie-Gauvin *et al.* 2017; Drouineau *et al.* 2018; Watson *et al.* 2018). Management of aquatic systems should be undertaken carefully and requires a well-grounded decision making process, whereby a framework is needed that incorporates impacts of altered water-mediated transport of abiotic (nutrients, sediment, toxins, energy) and biotic (invasive species) components (Pringle 2006). Ecological considerations to take into account include preventing or limiting range expansion of invasive species, preventing the spread of diseases, preventing individuals



from entering ecological traps and preventing genetic mixing (hybridization) of wild and stocked (hatchery-origin) populations, and these considerations should be balanced against the benefits of increased hydrologic connectivity (Jensen and Jones 2017; Milt *et al.* 2018). It is thus possible that, after analysis of such trade-offs of environmental effects in intensively developed aquatic systems, it may be preferential to maintain (or even further reduce) connectivity within waterways so as to minimize ecological impacts, especially in systems affected by, or with a history of, invasive species (Rahel 2013; Starrs *et al.* 2015). The ecological (and economic - Lovell *et al.* 2006) impacts of AIS on river systems have been extensively studied globally for a broad number of invasives (for example, Peterson *et al.* 2008; Shibata *et al.* 2011; Van der Walt *et al.* 2016; Pereira *et al.* 2017; Gobel *et al.* 2019), and warrant the need to be acknowledged as a top priority in catchment based management protocols (Wittmann *et al.* 2014; Pereira *et al.* 2018; Xiong *et al.* 2018).

2 ECOLOGICAL IMPACTS OF INVASIVE SPECIES

Biodiversity of freshwater ecosystems is in decline globally (Dudgeon *et al.* 2006), in part attributed to the consequences of biological invasions (Kitano 2004; Strayer and Dudgeon 2010; Simberloff and Vitule 2014), the spread of which has been facilitated considerably by human-induced actions including directly by connecting and opening of historically isolated waterways on large spatial scales, but also on small scales such as removal of river infrastructure, and indirectly by (long- distance) inter-basin transfers (on ship hulls, in ballast water; Leuven *et al.* 2009; Keller *et al.* 2011; Kelly *et al.* 2013). The invasion process is constituted of four stages; transport (displacement across major geographic barriers to invasion area, success dependent on survival rate of offspring), colonization (environmental conditions at the arrival site must be within tolerance range), establishment (crucial to survive natural enemies, potentially form mutualistic relationships and acquire resources) and landscape spread stage (expanding presence by establishing colony populations), each of which is crucial for a species to become invasive (Hellmann *et al.* 2008; Gallardo *et al.* 2016; Coulter *et al.* 2018). In freshwater, invasives often have generalized habitat requirements and may tolerate low dissolved oxygen concentrations and/or altered water quality, which facilitates their ability to displace and replace native fauna (Alonso and Castro-Diez 2008; Lodge *et al.* 2012). Thus, alterations to rivers, including damming, often generate habitat conditions, in which, a proportion of alien species colonise and spread to become AIS at the expense of native species, reducing the resilience of the ecosystem. Climate change may impact aquatic systems as well (Rahel *et al.* 2008), and may even potentially have a synergistic effect with ecological impacts by non-native species, because it may affect the distribution, spread, abundance, and impact of invasive species (Gritti *et al.* 2006; Hellmann *et al.* 2008; Roberts *et al.* 2017; Bae *et al.* 2018).

Multiple non-indigenous species may have an interactive effect on the community and ecosystem invaded (Ficetola *et al.* 2006; Johnson *et al.* 2009). Aside from the environmental impact, AIS may have an adverse effect directly to humans in the form of monetary losses, or may impact social structure (Charles and Dukes 2008). Ecological integrity may be affected through trophic cascades, whereby the newly introduced functional components (i.e. invasives) cause ecological impacts throughout the food chain (Strayer 2010) via top-down (Mercado-Silva *et al.* 2009) or bottom-up (Lodge *et al.* 2012; Heath *et al.* 2014) control of the food web. While direct biotic interactions between invader and indigenous species through predation, competition or grazing are more widely known (for example, Rösch *et al.* 2018; Gobel *et al.* 2019), alterations in habitat conditions may affect the recipient community as well (for example, Zaiko *et al.* 2007). Such habitat conditions (for example, nutrients or water clarity) may be changed dramatically and at high intensity by invasive ecosystem engineers, thereby affecting



population persistence of non-tolerant endemic species severely (Darrigran and Damborenea 2011; Sousa *et al.* 2014). While in some cases, AIS may have a perceived beneficial effect, for example zebra mussel *Dreissena* clearing eutrophic, turbid water (Caraco *et al.* 1997), negative impacts may still persist (for example, the commercial impact of fouling by zebra mussel).

Although the impacts of aquatic invasives, potentially constituting a broad spectrum of taxa, are widely recognized, freshwater managers often fail to address them sufficiently in order to protect regional biotic integrity (Tsuboi *et al.* 2013; Moorhouse and Macdonald 2015). One of the most efficient measures in managing AIS in order to control the spread of invasive species entails reducing the connectivity among habitats by barriers (Kates *et al.* 2012; Rahel 2013; Sherburne and Reinhardt 2016).

3 BARRIER MANAGEMENT TO EXCLUDE NON-NATIVES

Management of AIS requires an integrated and coordinated approach incorporating multiple elements. These include effective legislation and education to prevent the release of non-native species outside their natural range; predictive tools to classify and rank those species at greatest risk of becoming invaders; effective surveillance to identify invasions early on, and effective management measures to eradicate or control AIS (Koehn and MacKenzie 2004; Van der Zanden and Olden 2008; Hussner *et al.* 2017). River barriers have the potential to contribute towards management of the spread of AIS.

Where existing barriers occur within an invaded catchment, the distribution, type and permeability of those barriers form a key tool for the management of within-catchment invasion. Almost all river biota have one or more life stage which facilitate downstream movement, often by way of “drift” on water currents, and contribute to dispersal and recolonization (Kennedy *et al.* 2014; Lechner *et al.* 2014). Thus, the use of barriers to manage AIS is mostly limited to inhibiting their spread upstream. Since many aquatic plants and bacteria have passively drifting propagules that are effectively incapable of upstream movement, their independent spread upstream of a colonisation point is unlikely whether a barrier is present or absent immediately upstream. Instead, spread is likely to be via propagule contamination of vectors such as waterbirds (Figuerola *et al.* 2003) or boats (recreational or commercial; Donnelly and Walters 2008) and lentic habitats upstream of dams may provide conditions for multiple point-source colonisation and subsequent downstream spread from each. As such, it can be theorised that in impounded reaches upstream of the range of passively dispersing AIS, dam removal is more likely to inhibit spread of passive dispersing species, especially lacustrine ones. By contrast, the spatial distribution of barriers sufficient to prevent upstream dispersal and colonisation is increasingly applied in catchment management to inhibit upstream spread of actively dispersing animals. A GIS framework may be developed to assess management options within a catchment in combination with barrier network algorithms (King and O’Hanley 2016) so that the spread of invasives may be mapped and limited adequately.

At the early invasion stages, invasives should be prevented from dispersing to a new region or, once introduced, should be prevented from secondary spread from the introduction point, which may be achieved by strategic placement of barriers (Dana *et al.* 2011; Clarkson *et al.* 2012), especially for limiting natural spread further up the catchment. Even small barriers may potentially prove adequate in protecting local endemic communities (for example, Pratt *et al.* 2009; Rosewarne *et al.* 2013; Miehl *et al.* 2017). A common approach when invasive species have such strong biotic or abiotic interactions

with the receiving community that coexistence is unrealistic, for example,, the use of intentional fragmentation, whereby a barrier is constructed, non-native taxa are removed upstream of the barrier (if they have already arrived there) while native species are returned to and/or stocked in upstream reaches (Neeson *et al.* 2015). This strategy, also known as isolation management, is particularly relevant and needed when a non-native species has reached the establishment invasion phase when elimination has become unfeasible, and the focus for freshwater managers changes to population control (Fausch *et al.* 2009). When barriers are constructed, novel habitat may be created, which can be beneficial for invasives to colonize (Liew *et al.* 2016). Upstream of barriers, the conversion of natural lotic segments into artificial lentic habitat may impact rheophilic species which are intolerant to drastic changes in environmental conditions, after which invasives may fill these new niches (Liu *et al.* 2012; Vitule *et al.* 2012). Non-specialist habitat requirements and a tolerance for changed water quality, altered water temperature regimes or low dissolved oxygen may also explain the increasing presence of non-native species in tailwater areas of dams (Linares *et al.* 2018; Pereira *et al.* 2018; Xiong *et al.* 2018). Indeed, part of the argument made for the removal of barriers in restoration of rivers with non-natives is that it may restore conditions to those closer to which native species are adapted and enhance their ability to compete favourably with non-natives (Stanley and Doyle 2003; Fausch *et al.* 2009). A careful approach, as shown in **Figure 1**, should therefore be taken by freshwater managers when the decision is made to construct barriers.

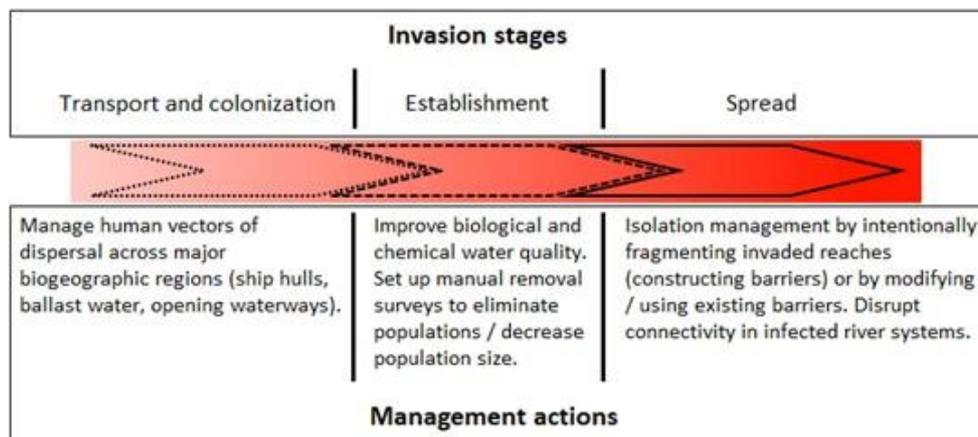


Figure 1: Invasion stages and possible management actions during the process of an aquatic invasion.

Reducing hydrologic connectivity in river systems is known to adversely affect animal populations (for example, Magilligan *et al.* 2016; Silva *et al.* 2018; Liu and Wang 2018). Ideally, intentional fragmentation should still allow for free movement of native biota, while blocking or limiting passage of undesirable, invasive species (for example, Starrs *et al.* 2015; Sherburne and Reinhardt 2016). This concept of selective passage has only recently received more attention (for example, McLaughlin *et al.* 2013; Moran-Lopez and Tolosa 2017), whereas historically biota (usually fish) passage efforts were almost always focused on facilitating migration of juveniles of target species to nursery areas and adults to spawning habitat (Clay 1995). For fish, a small portion of each population may move relatively long distances (Radinger and Wolter 2014), meaning invasive species may be able to rapidly spread to new areas if connectivity is not adequately reduced. Achieving well-functioning selective passage over barriers is dependent on the differential ability of species to meet, or fail to meet, one or more ecological criteria designed to filter out invasive species from desirable species in impacted reaches (Pellicice *et al.* 2015; Kirk *et al.* 2016). Biological traits, which determine the capacity for



organisms to move over or through a barrier (for example, physical, physiological or sensory barrier; Noatch and Suski 2012) are used as focal points for ecological filters (Mims and Olden 2013; Bajer *et al.* 2015).

4 SELECTIVE PASSAGE OF ANIMALS OVER BARRIERS BASED ON BIOLOGICAL TRAITS

The following sections refer to design principles for developing selective passage for animals, and emphasize control of active upstream movement as part of intentional isolation management methods. Most of the principles are illustrated by reference to fish, for which, the most and best examples are known (however, the broad principles also are relevant to other river animal biota). Although most of the examples currently available relate to limiting upstream movement, the principles can also be applied to lateral connections and downstream movement, even though for the latter, “drift” dispersal and colonization especially, makes this more difficult.

4.1 Physical attributes

Barriers may work effectively by selecting on biota locomotion, which includes swimming, jumping and climbing. Swimming capacity balances between endurance and speed, and multiple swimming speed categories exist. In increasing velocity, very long distance aerobic swimming at optimal swimming speed (U_{opt}), critical swimming speed (U_{crit}) with maximum swimming oxygen uptake and finally anaerobic burst swimming which can only be used for up to several seconds, which is defined as maximum swimming speed (U_{max}) (Videler 1993; Tudorache *et al.* 2008; Guderley and Pörtner 2010; Kieffer 2010). While flow regimes over barriers and through fishways may be complex with, for example, the occurrence of high turbulence under moderate to high discharge levels and low velocities observed in channel margins even under high flow conditions, structures such as fishways may be constructed so that the combination of flow velocities and fishway or barrier length may filter out invasive species in upstream reaches. There are some species which can withstand higher current velocities by employing a low-energy recovery strategy of holding in place, for it to them resume upstream ascent with partially replenished energy levels. Examples include lamprey (*Petromyzontiformes*), able to attach to surfaces employing a burst-attach-rest behaviour (Vowles *et al.* 2017), and non-native European round gobies (*Neogobius melanostomus*) that use their pectoral and modified pelvic fins to hold station at higher current velocities (Tierney *et al.* 2011).

Jumping ability of aquatic biota (almost exclusively fish) constitutes another selective filter, as fish differ in their maximum jumping height, dependant on several factors including species, body size and water temperature (Ficke and Myrick 2011; Baudoin *et al.* 2015). A barrier which requires fish to jump to traverse it, may thus filter out certain species which have a lower jumping ability. Perhaps the best known example of this is the use of lipped weirs in the Laurentian Great Lakes to prevent upstream passage of non-leaping sea lamprey (*Petromyzon marinus*), while allowing passage of jumping salmonids, but unfortunately not the passage of many non-jumping native migrants such as many cyprinids and catostomids (Pratt *et al.* 2009). Such approaches require an understanding of the jumping ability of native and non-native species. For example, (Holthe *et al.* 2005) defined the obstacle height necessary to prevent successful jumping by invasive Eurasian minnow (*Phoxinus phoxinus*) in Norwegian rivers dominated by salmonids. Common carp (*Cyprinus carpio*) are not conventionally known for their leaping ability, but in



Australia, this species was effectively filtered out from endemic species in confinement traps placed at dams and weirs by exploiting their unique jumping behaviour with a cage device (Stuart *et al.* 2006).

Climbing is a behavioural technique used by Pacific lamprey (*Entosphenus tridentatus*) in North America, as reported in Fricket *et al.* (2017). This species attaches to substrates with its sucker mouth, flexes its body in a wave form, then extends upward and reattaches. Passage success was high (94%) over a 1.6m high aluminium wetted weir. As the capacity to use climbing behaviour to overcome barriers is rare among fish, (Zhu *et al.* 2011), though it is widely adopted in anguilliform eels, it has the potential to be used as a trait for selective passage (Goodman and Reid 2017). Limited walking and climbing ability of invertebrates without an aerial life stage as a potential option has been proposed for crayfish (Kerby *et al.* 2005; Dana *et al.* 2011; Rosewarne *et al.* 2013), based upon reduced propensity to climb over smooth surfaces with low friction and/or high velocities and/or over vertical or lipped barriers. Fringset *et al.* (2013) found that signal crayfish (*Pacifastacus leniusculus*) employed short swimming events as well as walking to facilitate passage, requiring a combination of steep gradient and sufficient flow velocity to prevent passage, yet other studies have not reported obstacle passage by swimming in crayfish, and the importance of swimming in passing obstacles in nature has probably been overstated by Fringset *et al.* (2013). It has been suggested that existing river barriers may be ineffective in blocking the upstream spread of decapod crustaceans because many species can temporarily leave the water and walk around the barrier, but this has not been demonstrated other than in anecdotal circumstances, and is dependent upon the barrier size, channel edge characteristics and hydrologic conditions.

4.2 Physiological attributes

As species have different optimum temperatures, progressive warming of aquatic habitats may cause shifts in phenology, distribution and abundance of invasive and native species by limiting oxygen transport systems when approaching maximum aerobic scope (for example, Norin *et al.* 2014; Hesselschwerdt and Wantzen 2018). Animal taxa are regularly classified into thermal guilds, and temperature tolerance may be used in controlling invasive species (Null *et al.* 2013). Invasives falling into warm water adapted species, may be prevented from entering reaches if water temperatures are low enough and vice versa (Kernan 2015), so that thermal tolerance may be used as an ecological filter in blocking invasive species (Hesselschwerdt and Wantzen 2018).

Reducing the oxygen concentration in the water column, or hypoxia, is another strategy working on the physiological state of organisms which may be used to limit the spread of invasives (Domenici *et al.* 2013). There is also evidence of potentially increasing oxygen concentrations to above normal levels (hyperoxia), which may be used as a control trait (Wheatly 1989). Management of reservoir water quality through lowering oxygen content and lowering pH has been used as a successful management measure for achieving high mortality of invasive molluscs such as the Asiatic clam (*Corbicula fluminea*; Wittmann *et al.* 2012), and to slow population growth in crustaceans such as signal crayfish (*Pacifastacus leniusculus*; Wheatly 1989) and virile crayfish (*Orconectes virilis*; France 1993), but this has not acted as an absolute barrier (Sousa *et al.* 2014).

Exploiting the sensitivity of AIS to supersaturated concentrations of carbon dioxide may effectively limit the spread of AIS (Cupp *et al.* 2018; Schneider *et al.* 2018); Kates *et al.* (2012) evaluated the effectiveness on increased CO₂ concentrations in water for deterring fish movement, for invasive species (silver carp (*Hypophthalmichthys molitrix*) and bighead carp (*Hypophthalmichthys nobilis*)), and native (to North America) species (largemouth bass (*Micropterus salmoides*) and bluegill (*Lepomis macrochirus*)). It was concluded that zones of elevated CO₂ have potential to deter the movement of fish, though native species were also, but to a lesser degree, impacted. Similar impacts of elevated CO₂ were reported by Manea and



Leishman (2011). This wider, non-target species effect was not found for adult freshwater mussels (*Fusconaia flava*), which were found to experience limited physiological impacts from elevated CO₂ levels (Hannan *et al.* 2016).

4.3 Morphological attributes

In New Zealand, Daniel *et al.* (2014) evaluated the effectiveness of using a one-way fish barrier for common carp. Horizontal bars of 30mm bar spacing, successfully prohibited adult carp and eels from moving upstream past the barrier. Although juvenile invasive carp were able to pass upstream, it was deemed impractical to design a barrier capable of blocking all invasives, considering the conflicting need to provide free passage for native species. Furthermore, in a laboratory trial, selective passage was shown for common carp (Thwaites *et al.* 2010). When confronted by barriers, carp often attempt to jump over or push through them. Considering their relatively strong body morphology, a push trap element exploiting the pushing capacity of carp selectively filtered out invasive carp from native fish (Thwaites *et al.* 2010).

4.4 Behavioural attributes

The position in the water column at which fish swim may be used to selectively pass or exclude fish. Bottom orifices may selectively pass bottom dwelling fish (to potentially capture or remove them from the system, Jager *et al.* 2016). Selective passage based on diel activity has potential when managing AIS as barriers, as species may be distinctively nocturnal or diurnal (Johnson *et al.* 2016).

4.5 Sensory attributes

Sound may be used as a source for deterring or guiding invasive species in impacted water systems (Noatch and Suski 2012). Silver carp were shown to keep away from complex broadband sound sources, so that such acoustics may be used to selectively pass this species (Vetter *et al.* 2015). In a laboratory experiment, Zielinski *et al.* (2014) demonstrated the effectiveness of bubble curtains (of three different levels of coarseness) on common carp movement behavior. In the two course bubble trials, movement across the curtain was reduced by 75-85% in both directions, whereby, the different effect was the result of fluid motion and sound rather than visual cues. Invasive sea lamprey were found to be limited in their upstream movement by the use of low-frequency acoustics, strobe lights and a bubble curtain (Miehls *et al.* 2017).

Another sensory attribute which may be exploited in invasive species management is electricity. Electroreceptive organs are present on the heads, or as part of the lateral line system, in cartilaginous fish, teleosts and often in amphibians, and is used for locating prey (Northcutt *et al.* 1994). Overstimulate the electroreceptors by use of strong magnetic fields, and a nonphysical barrier may cause a repellent effect, thereby allowing selective passage (Gibbs and Northcutt 2004; Bajer *et al.* 2018). Spinycheek crayfish (*Orconectes limosus*) movement was found to be blocked by installing an electricity barrier (Benejam *et al.* 2015). Similar findings of electric barrier effectiveness on invasive species movement were documented by O'Farrell *et al.* (2014) and Johnson *et al.* (2016).

Olfactory sense has the potential to be used to filter out invasive species, as in the case of the invasive sea lamprey, which may be attracted toward, or lured away by, pheromone plumes (Meckley *et al.* 2014; Hume *et al.* 2015; Sorensen 2015; Wagner *et al.* 2018). Based on a controlled field trial in Australia, Hagman and Shine (2009) reported that alarm pheromones secreted by injured larval cane toads caused a reduction in tadpole survival rates, and tadpoles under repeated exposure to the pheromone did not



seek shelter. The authors recognize larval pheromones as a means to control cane toad recruitment in nature, although more research is needed. From a field test, Wagner *et al.* 2006 demonstrated the effect of a migratory pheromone in luring sea lamprey toward a pheromone-baited trap. A high proportion (90%) of migrating individuals were drawn to pheromone-treated streams.

Application of metal as a contact deterrent substrate barrier has been tested to block the spread of the invasive New Zealand mudsnail (*Potamopyrgus antipodarum*; Hoyer and Myrick 2012). Copper-based substrates were shown to limit locomotor activity of the mudsnail under a set of water velocities and temperatures, pH values, hardness levels and levels of surface fouling.

To summarize, physical and non-physical barriers to free movement of invasive aquatic biota come in many forms and vary in effectiveness. Physical barriers may not always keep their impact on the fitness of native species at a low flow rate (for example, injury from exclusion barscreens), whereas non-physical barriers typically cause less injury and do not hinder navigation or constrain water flow (Noatch and Suski 2012). If a barrier is determined to be ineffective for managing alien species, combining multiple types of barriers may result in an arrangement to greatly limit or even stop the spread of invasive species (for example, McLaughlin *et al.* 2007; Miehl *et al.* 2017) which would benefit freshwater managers.

5 INVASIVE AQUATIC BIOTA IN EUROPE: IMPACTS AND CONTROL

Effective management of freshwater invasive biota requires knowledge of their abundance and rate of spread at appropriate spatial and temporal scales. For this report, invasive species in Europe are presented according to their impact (**Table 1**). A semi-quantitative tool (generic impact scoring system (GISS)) was used which reflects evidence of impact of invasive species. Twelve categories are distinguished, each ranging from 0 (no impact detectable) to 5 the highest impact possible) for a total possible score of 60 (highest impact; Nentwig *et al.* 2016; 2018). The table summarizes the potential for using physical and non-physical barriers for managing the spread of these invasive fish, together with references to further information. Inevitably, not all species are included in the listing since the perceived impact of different species varies from locality to locality and is changing with time. For example, the wels catfish (*Silurus glanis*) is native to some parts of Europe, but has been introduced to other parts of Europe, often for recreational angling. Where it has been introduced and is successfully breeding, it has become a major, non-native apex predator causing concern as to its impact. It has been introduced to many reservoirs above dams, which provides a suitable habitat, but in the River Garonne in South Western France, individuals have also been found to inhabit fishway pools predated migrant species (Boulêtreau *et al.* 2018). Although some AIS, such as common carp, are widely naturalised, they are non-native and widely acknowledged for the extent of their impact in floodplain systems (Vilizzi *et al.* 2015).



Table 1: Freshwater invasive species in EU, listed are up to 10 species per taxonomic group with highest ecological impact, based on the top 100 EU invasive species list as presented in Nentwig et al. 2018. Species are ordered from high to low impact based on the generic impact scoring system (GISS). Reptilia are not included in the table due to low scores on the GISS system.

Taxonomic group	Family	Total impact (GISS score; [0 – 60])	Type of barrier and condition acted on. Barrier types potentially linked to physical barriers such as dams, weirs and their management (for example, water quality, sensory characters are in bold)	References
Plants				
<i>Heracleum mantegazzianum</i> – Giant hogweed	Apiaceae	24	Chemical barrier (herbicidal control zones) ^{a,b}	Caffrey and Madsen 2001 ^a ; Nielsen et al. 2005 ^b
<i>Crassula helmsii</i> – New Zealand pygmy weed	Crassulaceae	22	Physiological barrier (inundation with seawater^{a,b}; changes in water temperature^c)	Charlton et al. 2010 ^a ; Gardiner and Charlton 2012 ^b ; Anderson et al. 2015 ^c
<i>Fallopia japonica</i> – Japanese knotweed	Polygonaceae	21	Chemical barrier (phloem-mobile herbicides control zones) ^a	Bashtanova et al. 2009 ^a
<i>Hydrocotyle ranunculoides</i> – Floating pennywort	Araliaceae	19	Chemical barrier (herbicide ^a), Physiological barrier (draining^b; changes in water temperature^c)	Newman and Dawson 1999 ^a ; Hussner and Meyer 2009 ^b ; Anderson et al. 2015 ^c
<i>Ludwigia grandiflora</i> – Water primrose	Onagraceae	19	Chemical barrier (herbicidal control zones) ^a	Sartain et al. 2015 ^a
<i>Ludwigia peploides</i> – Creeping water primrose	Onagraceae	19	Chemical barrier (herbicidal control zones) ^{a,b}	Richardson et al. 2008 ^a ; Saunders and Pezeshki 2014 ^b
<i>Baccharis halimifolia</i> – Groundsel tree	Asteraceae	17	Physiological barrier (alteration of hydrodynamic regime)	Caño et al. 2013; Frau et al. 2014 ^a
<i>Solidago canadensis</i> – Goldenrod	Asteraceae	17	Physiological barrier (alteration of light intensities)	Sun et al. 2008 ^a
<i>Impatiens glandulifera</i> – Himalayan balsam	Balsaminaceae	9	Physiological barrier (alteration of hydrology – flooding events^a; changes in water temperature^b)	Tickner et al. 2001 ^a ; Willis and Hulme 2002 ^b
<i>Myriophyllum aquaticum</i> – Parrot's feather	Haloragaceae	6	Physiological barrier (changes in water temperature)^a	Anderson et al. 2015 ^a
<i>Lysichiton americanus</i> – American skunk cabbage	Araceae			Schrader et al. 2010



Animalia				
Mollusca				
<i>Dreissena polymorpha</i> – Zebra mussel	Dreissenidae	23	Physical barrier (filtering, flow velocity) ^a ; Physiological barrier (increased water temperature) ^b ; Chemical barrier (control zones (three molluscicides)) ^c	Bodamer and Bossenbroek 2008 ^a ; Anderson <i>et al.</i> 2015 ^b ; Lund <i>et al.</i> 2018 ^c
<i>Pomacea canaliculata</i> – Golden apple snail	Ampullariidae	20	Chemical barrier (molluscicide control zones) ^a ; Physiological barrier (changes in water temperature ^{b,c} ; lower pH, salinity ^d)	Wada 2004 ^a ; Matsukura <i>et al.</i> 2009 ^b ; Yoshida <i>et al.</i> 2014 ^c ; Bernatis <i>et al.</i> 2016 ^d
<i>Potamopyrgus antipodarum</i> – New Zealand mud snail	Hydrobiidae	19	Physiological barrier (different temperatures ^a ; changes in endocrinology ^c ; increased CO ₂ concentrations ^d); Chemical barrier (molluscicide control zones) ^b	Richardson <i>et al.</i> 2004 ^a ; Gust <i>et al.</i> 2009 ^b ; Gust <i>et al.</i> 2010 ^c ; Treanor <i>et al.</i> 2017 ^d
<i>Pomacea maculata</i> – Island applesnail	Ampullariidae	17	Physiological barrier (lower water temperature ^a ; lower pH, salinity ^b)	Yoshida <i>et al.</i> 2014 ^a ; Bernatis <i>et al.</i> 2016 ^b
<i>Marisa cornuarietis</i> – Colombian ramshorn apple snail	Ampullariidae	14	Physiological barrier (lower water temperature ^a); Chemical barrier (molluscicide control zones) ^b	Selck <i>et al.</i> 2006 ^a ; Forbes <i>et al.</i> 2008 ^b
<i>Corbicula fluminea</i> – Asiatic clam	Corbiculidae	10	Chemical barrier (control zones of halogens) ^a ; Physiological barrier (gas impermeable benthic barriers to reduce dissolved oxygen (DO) concentrations ^b ; increased CO ₂ concentrations ^c)	Doherty <i>et al.</i> 1986 ^a ; Wittmann <i>et al.</i> 2012 ^b ; Treanor <i>et al.</i> 2017 ^c
Arthropoda				
Crustacea				
<i>Dikerogammarus villosus</i> – Killer shrimp	Gammaridae	16	Physiological barrier (increased water temperature) ^{a,c} ; Physical barrier (height, flow velocity) ^b	Maazouzi <i>et al.</i> 2011 ^a ; Macneil and Platvoet 2013 ^b ; Anderson <i>et al.</i> 2015 ^c
<i>Procambarus clarkii</i> – Red swamp crayfish	Cambaridae	34	Physical barrier (height, flow velocity) ^{a,b}	Kerby <i>et al.</i> 2005 ^a ; Frings <i>et al.</i> 2013 ^b
<i>Eriocheir sinensis</i> – Chinese mitten crab	Varunidae	28	Physiological barrier (changes in water temperature, salinity) ^a ; Physical barrier (height, flow velocity) ^b	Anger 1991 ^a ; Frings <i>et al.</i> 2013 ^b
<i>Procambarus fallax</i> – Georgia crawfish	Cambaridae	28	Physical barrier (height, flow velocity) ^a ; Chemical barrier (pesticidal control zones) ^b	Frings <i>et al.</i> 2013 ^a ; Koutnik <i>et al.</i> 2017 ^b
<i>Pacifastacus leniusculus</i> – Signal crayfish	Astacidae	19	Physiological barrier (zones of hyperoxia) ^a ; Physical barrier	Wheatly 1989 ^a ; Bubb <i>et al.</i> 2005 ^b ; Frings <i>et al.</i> 2013 ^c

			(height, flow velocity) ^{b,c}	
<i>Cherax quadricarinatus</i> – Redclaw crayfish	Parastacidae	18	Chemical barrier (pesticidal control zones) ^a ; Physical barrier (height, flow velocity)^b	Frontera <i>et al.</i> 2011 ^a ; Frings <i>et al.</i> 2013 ^b
<i>Cherax destructor</i> – Common yabby	Parastacidae	16	Physiological barrier (reduced water temperature)^a; Physical barrier (height, flow velocity)^b	Verhoef <i>et al.</i> 1998 ^a ; Frings <i>et al.</i> 2013 ^b
<i>Orconectes virilis</i> – Virile crayfish	Astacidae	15	Physiological barrier (reduced pH)^a; Physical barrier (height, flow velocity)^b	France 1993 ^a ; Frings <i>et al.</i> 2013 ^b
<i>Orconectes limosus</i> – Spinycheek crayfish	Astacidae	8	Physical barrier (height, flow velocity)^a; Sensory barrier (electrical current)^b	Frings <i>et al.</i> 2013 ^a ; Benejam <i>et al.</i> 2015 ^b
Chordata				
Fish				
<i>Carassius auratus</i> – Goldfish	Cyprinidae	24	Sensory barrier (olfactory)^a; Physical barrier (height, jumping capacity)^b	Sorensen and Stacey 2004 ^a ; Morán-López and Tolosa 2017 ^b
<i>Oreochromis mossambicus</i> – Mozambique tilapia	Cichlidae	21	Sensory barrier (olfactory)^a; Physical barrier (swimming capacity)^b	Sorensen and Stacey 2004 ^a ; Starrs <i>et al.</i> 2017 ^b
<i>Oreochromis niloticus</i> – Nile tilapia	Cichlidae	18	Sensory barrier (olfactory)^a; Physical barrier (height, flow velocity)^b; Physiological barrier (increased CO2 concentrations)^c	Sorensen and Stacey 2004 ^a ; Zambrano <i>et al.</i> 2006 ^b ; Treanor <i>et al.</i> 2017 ^c
<i>Pseudorasbora parva</i> – Topmouth gudgeon	Cyprinidae	17	Sensory barrier (olfactory)^a; Physical barrier (height, flow velocity)^b	Sorensen and Stacey 2004 ^a ; Chu <i>et al.</i> 2015 ^b
<i>Oncorhynchus mykiss</i> – Rainbow trout	Salmonidae	17	Sensory barrier (olfactory)^a; Physiological barrier (increased CO2 concentrations)^b; Physical barrier (swimming capacity)^c	Sorensen and Stacey 2004 ^a ; Clingerman <i>et al.</i> 2007 ^b ; Starrs <i>et al.</i> 2017 ^c
<i>Micropterus dolomieu</i> – Smallmouth bass	Centrarchidae	16	Sensory barrier (electrical)^a; Physiological barrier (reduced water temperature)^b	Verrill and Berry 1995 ^a ; Landsman <i>et al.</i> 2011 ^b
<i>Ctenopharyngodon idella</i> – Grass carp	Cyprinidae	16	Physical barrier (height, flow velocity)^a	Maceina <i>et al.</i> 1999 ^a
<i>Poecilia reticulata</i> – Guppy	Poeciliidae	13	Chemical barrier (pesticidal control zones (atrazine)) ^a	Araújo <i>et al.</i> 2018 ^a
<i>Gambusia holbrooki</i> – Eastern mosquitofish	Poeciliidae	11	Chemical barrier (pesticidal control zones) ^a ; Physical barrier (swimming capacity)^b; Physiological barrier (increased CO2 concentrations)^c	Willis and Ling 2000 ^a ; Starrs <i>et al.</i> 2017 ^b ; Treanor <i>et al.</i> 2017 ^c
<i>Phoxinus phoxinus</i> – Eurasian minnow	Cyprinidae		Sensory barrier (olfactory)^a; Physical barrier (height, jumping capacity, flow velocity)^{b,c}	Sorensen and Stacey 2004 ^a ; Holthe <i>et al.</i> 2005 ^b ; Kukuła <i>et al.</i> 2006 ^c



<i>Cyprinus carpio</i> – Common carp	Cyprinidae		Sensory barrier (electrical^{a,c,d}; olfactory^b); Physical barrier (height, jumping capacity^e; body morphology (bar grates)^f); Physiological barrier (increased CO2 concentrations)^{g,h}	VerrillandBerry1995 ^a ;Sorensenand Stacey 2004 ^b ; Dettmers <i>et al.</i> 2005 ^c ; Parker <i>et al.</i> 2015 ^d ;Holtheet <i>al.</i> 2005 ^e ; Hillyard <i>et al.</i> 2010 ^f ; Donaldson <i>et al.</i> 2016 ^g ; Treanor <i>et al.</i> 2017 ^h
Amphibia				
<i>Lithobates catesbeianus</i> – American bullfrog	Ranidae	18	Physical / physiological barrier (managing waterbodies to limit reproduction)^a; Physiological barrier (increased CO2 concentrations)^b	Madalozzo <i>et al.</i> 2016 ^a ; Treanor <i>et al.</i> 2017 ^b
<i>Xenopus laevis</i> – African clawed frog	Pipidae	15	Physical barrier (height, increased flow velocity and discharge)^a; Chemical barrier (insecticidal control zones)^b; Physiological barrier (increased CO2 concentrations)^c	Loboset <i>al.</i> 2013 ^a ;Yu <i>et al.</i> 2013 ^b ; Treanor <i>et al.</i> 2017 ^c



The current extent of AIS management with barriers is quite crude because most existing barriers to AIS also impact a variety of native species. In the future, as selective passage techniques evolve, effectiveness of management programmes for freshwater invasive species will be more dependent on supporting selective passage, so that desirable, endemic species are left unhindered in their movement, while, at the same time, unwanted, invasive biota are blocked from colonizing areas upstream or downstream. Based on this concept, examples of management strategies of an invasive species indicative of each taxonomic group are discussed in the next section.

Floating pennywort (*Hydrocotyle ranunculoides*), is native to North America and was naturalized in South and Central America, but it is considered invasive in western Europe. Its main habitat is stagnant or lentic waters such as canals, lakes and ponds, where it can quickly grow to form dense interwoven mats under the right environmental conditions (sunlight and high nutrient availability), thereby blocking sunlight into the water column. Since physical barriers cause impounded zones upstream, settlement of invasive species such as *H. ranunculoides* may be facilitated through unintentional transfer of propagules (Hoeinghaus *et al.* 2009; Casimiro *et al.* 2017). Hussner and Meyer (2009) investigated the response by floating pennywort at three different water levels, and found that growth rate and chlorophyll content was hindered by drained and semi-drained conditions compared to waterlogged habitat. Changes in the hydrologic regime by closely managing operating regimes of sluices and levees, or (temporarily) altering existing physical structures so a different hydrologic regime can be achieved in reaches invade by floating pennywort, may thus be beneficial in controlling its spread (**Table 1**).

The crustacean *Pacifastacus leniusculus*, also known as signal crayfish, is endemic to western North America, but since the 1960s, has been introduced to over 20 countries in Europe. It has caused, and is causing, further decline of European crayfish species by its continued spread within and between catchments (Bubbet *et al.* 2005; Ruokonen *et al.* 2018). Physical barriers have been used as a management strategy to control the spread of signal crayfish (**Table 1**) Fringset *et al.* (2013) studied the effectiveness of physical barriers to block passage of *P. leniusculus*, while still allowing passage of fish species, and concluded that physical barriers can be successful in protecting native crayfish at obstacles/passage devices with sufficiently high flow velocities and gradients. Weirs and substantial natural waterfalls may also prevent upstream colonization by undesired crayfish species, as argued by Bubbet *et al.* (2005). In catchments where signal crayfish have been encountered, modifications to existing barriers to increase flow velocities and gradients may limit the spread of this invasive. Such restoration efforts should be undertaken with careful consideration of endemic biota in the invaded river system, so as not to impede on their free movement.

The African clawed frog *Xenopus laevis*, originates from sub-Saharan Africa and is a successful amphibian invader which has spread across many major areas globally. Although dispersed widely, it has not been perceived as a species that causes impacts on biodiversity, but has been acknowledged as a vector for a fungus which may cause amphibian decline (Poundset *et al.* 2006). While techniques aimed at controlling the spread of the African clawed frog exist, including draining wetland habitats to remove habitat essential for reproduction (Maret *et al.* 2006), they are often ineffective or unfeasible (Snow and Witmer 2010). They may have negative effects on non-target species, as reported by Yu *et al.* (2013), for example, insecticides which caused mortality, malformations and growth impairments for not only *X. laevis*, but for other endemic amphibians as well. An emerging management approach is using physiological barriers of elevated CO₂ concentrations in the water column (**Table 1**), which have been reported to cause 100% mortality of *X. laevis* tadpoles at sufficiently high concentrations (Treanor *et al.* 2017). An adaptive barrier management approach may be developed, such as GIS framework to evaluate the distribution of physical barriers in the invaded catchment in combination with the use of barrier network algorithms to simulate fragmentation metrics (King and O'Hanley 2016). At sites where physical barriers



are present, and where the invasion front has established, barriers may be modified to enable infusion of CO₂ into the water column. The combination of different types of barriers acting on different biological traits of this invasive species (the in-stream physical barrier plus the physiological barrier) may prove successful in stopping their spread.

The Eurasian minnow is a small-bodied cyprinid and is considered invasive in some parts of Europe, principally parts of Scandinavia in areas in which benthic macroinvertebrate diversity has declined and a marked increase in small zoobenthos has been recorded (Borgstrøm *et al.* 1996). Indigenous fish of temperate river systems such as salmonids, share food and habitat niche space with the European minnow, and interspecific competition may thus be prevalent (Museth *et al.* 2010). Endemic fish species such as brown trout (*Salmo trutta*) have become impacted in their recruitment, yield and fitness as a result of minnow expansion across waterways (Museth *et al.* 2007). By using artificial barriers, the often undesired Eurasian minnow can be selectively prevented from reaching upstream reaches based on their limited jumping capacity (**Table 1**), while still allowing salmonids, such as brown trout, to pass, as they possess a stronger capacity to jump. Holthe *et al.* (2005) showed that further unwanted dispersal of the minnow was prevented by constructing suitable waterfall barriers, while brown trout were still able to pass the barrier. The effect was temperature dependent; for a water temperature of 4.9-6.5 °C, no successful leap of minnows was recorded, even for a barrier of only 3cm in length, while brown trout leaped over barriers up to 40cm in height. At 14.0-16.5 °C, minnows were able to negotiate waterfall barriers up to 27cm high. By making use of barriers in an identical approach, it may be possible to allow for selective passage of desired species while limiting the spread of undesired fish based upon their capacity to overcome vertical barriers.

6 CONCLUSION

Finding the right balance between protecting and conserving freshwater biodiversity on the one hand, and allowing unhindered movement up and downstream of endemic species while minimizing that of invasive species on the other, is crucial for effective management of waterways. Methods of selective passage are likely to become increasingly important to protect aquatic species richness (Silva *et al.* 2018), and the focus in water management is gradually changing from purely maximizing the biomass of economically important species (Rahel and McLaughlin 2018). Impacts of invasive species in river systems are dependent on characteristics of the invader and of the invaded ecosystem (Ricciardi and Atkinson 2004), and can manifest across multiple trophic levels (Maggi *et al.* 2015). The value of conservation of naturally occurring aquatic species in the system should therefore be analyzed on a case by case basis, with utmost consideration of the detrimental effect of invasive species if their spread is not blocked or limited by the uses of stream barriers, be they physical, physiological or chemical. Accordingly, when carrying out, or updating River Basin Management Plans under the Water Framework Directive, all European fresh water catchment managers should consider threats from AIS and the potential (or otherwise) for barriers to facilitate their control as part of an Adaptive Management Framework.



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