Preventing and controlling non-native species invasions to bend the curve of global freshwater biodiversity loss

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Abstract

The Emergency Recovery Plan for freshwater biodiversity recognises that addressing non-native species is one of six principal actions needed to bend the curve in freshwater biodiversity loss. This is because introduction rates of non-native species continue to accelerate globally and, where these species develop invasive populations, they can have severe impacts on freshwater biodiversity. The most effective management measure to protect freshwater biodiversity is to prevent introductions of non-native species. Should a non-native species be introduced, however, then its early detection and the implementation of rapid reaction measures can avoid it establishing and dispersing. If these measures are unsuccessful and the species becomes invasive then control and containment measures can minimise its further spread and impact. Minimizing further spread and impact includes control methods to reduce invader abundance and containment methods such as screening of invaded sites and strict biosecurity to avoid the invader dispersing to neighbouring basins. These management actions have benefitted from developments in invasion risk assessment that can prioritise species according to their invasion risk and, for species already invasive, ensure that management actions are commensurate with assessed risk. The successful management of freshwater non-native species still requires the overcoming of some implementation challenges, including non-native species often being a symptom of degraded habitats rather than the main driver of ecological change, and eradication methods often being non-species specific. Given the multiple anthropogenic stressors in freshwaters, non-native species management must work with other restoration strategies if it is to deliver the Emergency Recovery Plan for freshwater biodiversity.

1. <u>Introduction</u>

Freshwater ecosystems are subjected to considerable physical, chemical, and biological alteration through the exploitation of their provisioning ecosystem services, with these factors driving substantial declines in biodiversity (Tickner *et al.* 2020). One of these modifications is the introduction of non-native species (Moorhouse and Macdonald 2015; also see Box 1). Although only a proportion of the introduced species establish populations that then disperse, it is these invasive populations that can severely impact freshwater ecosystems across large spatial areas (Gozlan *et al.* 2010; Gallardo *et al.* 2016). Impacts of freshwater invasive species can manifest at levels from individual to ecosystem, and can include substantial declines in the diversity of native species (Cucherousset and Olden 2011; Flood *et al.* 2020) and altered ecosystem functioning (Vilizzi, Tarkan and Copp 2015), as well as causing major economic consequences (Cuthbert *et al.* 2021). Correspondingly, the 'Emergency Recovery Plan' of Tickner et al. (2020) recognised the successful management of non-native species as one of six principal actions needed to 'bend the curve' in freshwater biodiversity loss.

With the impacts of non-native species described by the International Union for Conservation of Nature as 'immense, insidious, and usually irreversible', it is not unexpected that management measures to restore invaded systems are challenging, with intensive efforts often needed for reducing the population abundances of non-native species and avoiding their further dispersal (Britton, Gozlan and Copp 2011a). Whilst efforts to manage non-native species in the wild have had some successes for fishes (Rytwinski *et al.* 2019) and macrophytes (Coetzee *et al.* 2021), the results for other invasive freshwater taxa have been more mixed, with the control of widely distributed invaders - such as non-native crayfish - being particularly difficult (e.g., Gherardi *et al.* 2011). Management control and containment efforts can also be resource intensive when applied over large spatial scales, thus pointing to the importance of preventing introductions as a key goal in the management of non-native species (Russell *et al.* 2017).

In the last decade, decision-making relating to the management of non-native species has been assisted by substantial developments in invasion risk assessments, with growing advancements in establishing minimum standards (e.g., Leung *et al.* 2012; Roy *et al.* 2018). Eradication feasibility assessment schemes have also helped prioritise the management of new and emerging invasive species (e.g., Booy *et al.* 2017; Booy *et al.* 2020). Thus, while invasive species continue to be a major driver of freshwater biodiversity decline, risk-based tools can enhance the prevention of high-risk invasive species being introduced and identify those species already present that require rapid management actions to prevent their invasion.

The primary aim of this study is to consider how managing non-native species can help 'bend the curve' in freshwater biodiversity loss within the Emergency Recovery Plan for freshwater biodiversity (Tickner *et al.* 2020). Through syntheses of existing knowledge, we identify the contemporary issues associated with non-native species in freshwaters, before outlining the strategies available for preventing, controlling, and coping with freshwater invasions (Fig. 1). We then discuss the integration of these tools into risk-based management programs, and outline the barriers to their successful implementation.

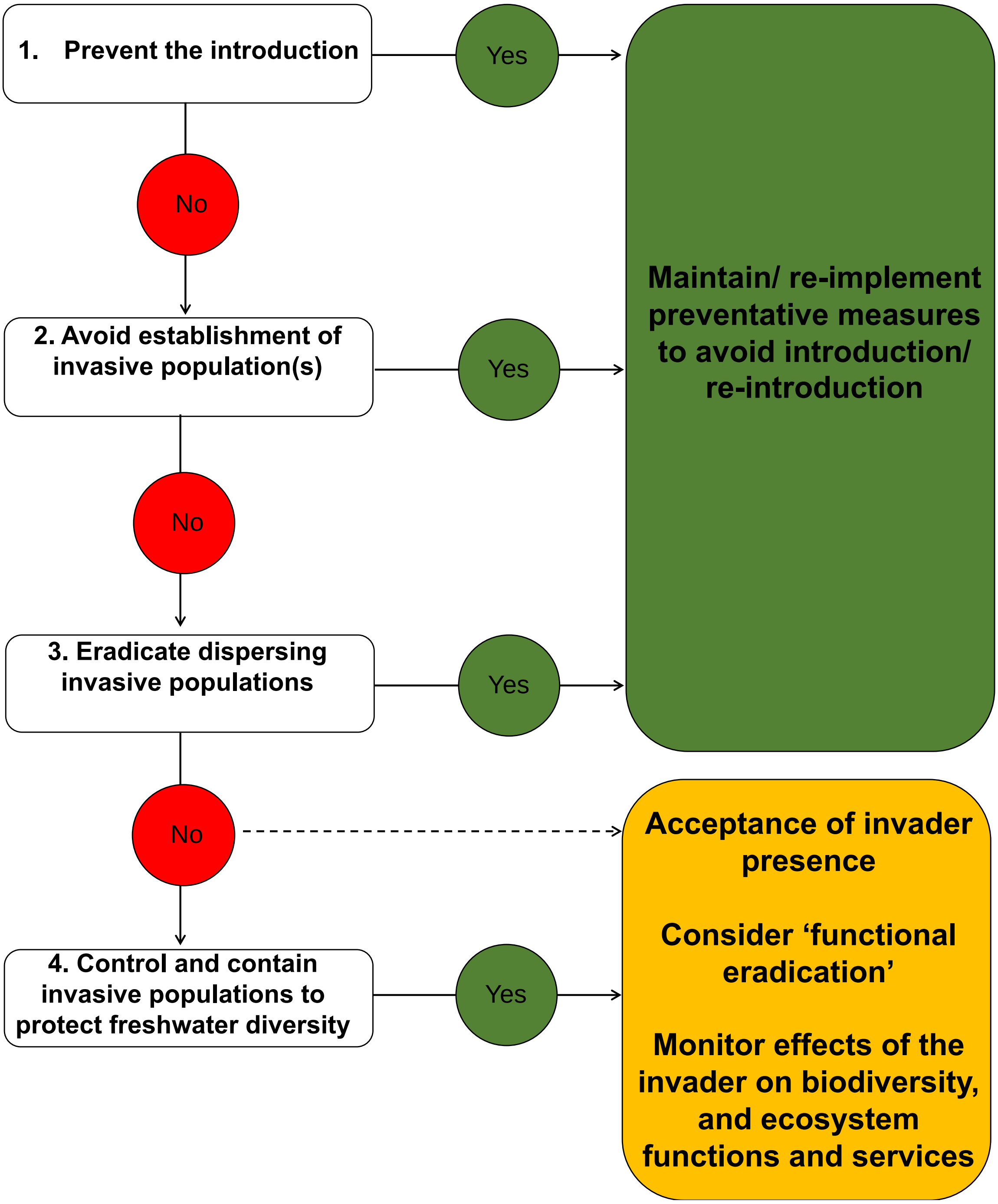

Figure 1. Step-wise process outlining the different actions to be considered for managing the impacts of a non-native species in freshwaters, where 'yes' indicates success of the measures underlying the action (see Table 1), 'no' indicates failure of the measures, solid arrows indicate primary pathway through the process, the dashed line indicates where no management interventions are taken against the presence of an invader. For information on functional eradication, see Green and Grosholtz (2021).

2. Non-native species in fresh waters

There are two major contemporary issues associated with non-native species in freshwaters: (i) their continued high rates of introductions; and (ii) for those introduced species, their ecological impacts on freshwater biodiversity.

2.1 Introduction rates

Major introduction pathways (i.e., the routes by which a species is transported from its native range to the new range; Saul *et al*. 2017) of freshwater non-native species vary taxonomically, and their strength and geographic routes have changed, and will continue to change over time. Primary motivations for early non-native species introductions were for extensive fish culture and stocking of plants and animals for the 'national good' (e.g., acclimatisation societies) (Hickley and Chare 2004). For example, initial introductions of common carp *Cyprinus carpio* to Western Europe occurred about 2000 years ago and were most likely facilitated by the Romans and later by Catholic monks who reared them in monastery ponds for food (Copp *et al.* 2005). Over the last 150 years, however, freshwater introduction rates have accelerated in association with the substantial increases in global trade, human population sizes, and tourism (Mormul *et al.* 2022). Since World War II, a more pronounced increase in the number of freshwater introductions has coincided with the shift towards more intense global trade and productivity (Seebens *et al.* 2017; Vitule *et al.* 2019). This increased number of introductions is consistent across continents and is projected to continue until at least 2050 (Seebens *et al.* 2021). As non-native freshwater species are widely used in the aquaculture and ornamental trade, high introduction rates are apparent across diverse taxa (e.g., fish, algae, crustaceans and molluscs; Dawson *et al.* 2017).

Contemporary introduction pathways for non-native species into freshwaters are diverse, although typically involve aquaculture practices sport angling, the ornamental trade, shipping and boating activities, cultural activities, and stocking for biocontrol (Dawson *et al.* 2017). Although many introductions of non-native species are intentional, especially where they are being used to increase food production and enhance sport angling, unintentional introductions are frequent, such as from ship ballast water releases, biosecurity lapses involving trade in ornamental pets, and the transfer of propagules attached to recreational boats that move between waterbodies (Gozlan *et al.* 2010; Mangiante *et al.* 2018). The ability of an introduced species to establish a population, disperse, and impact native biodiversity is elevated when the receiving freshwater has been extensively modified by anthropogenic activities, such as including the building of impoundments (promoting establishment) and canals (assisting dispersal) (Craig *et al.* 2017).

2.2 Ecological impacts

Freshwater invasive species can cause considerable negative ecological impacts through various processes (e.g., predation, competition, genetic introgression, pathogen transmission) that can manifest across different levels of organisation (i.e., genetic to ecosystem) and scales (i.e., local to global). The genetic impacts of aquatic invasive species include changes in genetic introgression leading to hybridization that can decrease genetic integrity in native populations through genetic pollution (e.g., hybrid swarms), as seen between non-native rainbow trout *Oncorhynchus mykiss* and native cutthroat trout *Oncorhynchus clarkii* subspecies and other salmonids in North American streams (Muhlfeld *et al.* 2009).

Population level impacts include changes in the abundance and distribution of native species and the transmission of pathogens and parasites. For instance, non-native crayfishes reduce the abundance of basal resources like aquatic macrophytes and aquatic invertebrates (e.g., snails, mayflies) through direct predation, and competition for habitat and prey with native crayfish, amphibians, and fish (Twardochleb, Olden and Larson 2013). Indeed, comparative functional responses (relationships between resource availability and resource uptake rate) have consistently revealed that invaders with high ecological impacts tend to have higher maximum consumption rates than trophically analogous natives (Dick *et al.* 2017), such as in invasive channel catfish *Ictalurus punctatus* versus native *Rhamdia quelen* in Brazil (Faria, Alexander and Vitule 2019). Invaders with lower consumption rates than native species are also often predicted to have substantial ecological impacts due to their relatively high population abundances, such as in topmouth gudgeon *Pseudorasbora parva* (Dick *et al.* 2017)

At the community level, freshwater invaders contribute to the local extinction of native species, and modify species composition and diversity. For example, the giant reed *Arundo donax* has invaded many Mediterranean-climate and subtropical riparian areas of the world, resulting in the decreased diversity of riparian vegetation that leads to lower abundance and diversity of riparian invertebrates and birds (Maceda-Veiga *et al.* 2016). The modification of species composition and diversity patterns through introductions has dramatically homogenised the present-day biogeography of the world's freshwaters, with distant regions now demonstrating striking similarities in their faunas and floras (Olden, Comte and Giam 2018).

Freshwater invaders can also alter ecosystems through 'ecosystem engineering' (Gallardo *et al.* 2016) where, for example, invasive bivalves (e.g., freshwater golden clam *Corbicula fluminea* and golden mussel *Limnoperna fortunei*) capture and consume suspended particles, produce faeces and pseudofaeces, function as an important resource subsidy, and bio-amplify pollutants throughout the food chain (Sousa *et al.* 2014). The high filtration rates of bivalves also typically reduce phytoplankton, increase water clarity, and thus change primary productivity and food web structure (e.g., shifts to more macrophytes) (Higgins and Vander Zanden 2010).

3. Managing freshwater non-native species

Managing freshwater non-native species involves preventing introductions (via mechanisms supported by horizon scanning, risk assessment and appropriate biosecurity at holding facilities), preventing introduced species from establishing and containing their secondary spread (via early detection and rapid response), and then controlling invasive populations to suppress their ecological impacts and reduce their rate of dispersal (Fig. 1). Where this step-wise process is unsuccessful then there is the option of living with the invader and, ideally, monitoring the consequences of this for freshwater biodiversity, and ecosystem functions and services (Fig. 1; Table 1).

3.1 Preventing new introductions

Effective strategies for preventing new species introductions involve the application of a series of tools and approaches, such as enforcement of strong legislation and regulation, horizon scanning and risk assessment, and the application of effective biosecurity measures coupled with education schemes.

Legislation and regulatory frameworks

The legislation and regulation of freshwater non-native species is largely reactive and implemented at national scales, despite the major pathways of introduction usually involving international transportation (Padilla and Williams 2004). An exception is the Ballast Water Management Convention (https://www.imo.org/), which aims to reduce the transfer and impact of aquatic organisms transported in the ballast water and sediment of ships (Gollasch *et al.* 2007). Several studies have investigated the capacity of ballast water exchange and treatment to remove the number of organisms (*cf.* Lakshmi, Priya and Achari 2021), and some success has been reported for the North American Great Lakes (Ricciardi and MacIsaac 2022). Yet its overall effectiveness at reducing the rate of introduction of new invasive species remains difficult to measure, especially with all aspects of the Convention yet to be fully implemented. Nevertheless, the Ballast Water Management Convention sets the example for the regulation of other major introduction pathways in fresh waters, notably the pet and aquaculture trade, for which 'safe' lists are a way to reduce intentional and accidental introductions, where such lists can be based on risk assessment processes (Padilla and Williams 2004).

At the pan-continental scale, the European Regulation 1143/2015 of Invasive Alien Species lists 30 freshwater species out of 88 'invasive alien species' as priorities for introduction prevention and invasion management. International trade is restricted for the species in the Union List that are not yet present in Europe; Member States must also design management plans adapted to their current levels of invasion. This legislation uses formal risk assessment processes for potentially invasive taxa (economic-, environmental-, and disease-focused) (CIRCABC 2022). The use of such approaches can result in the development of statutory lists (Roy *et al.* 2018), where 'black' and 'white' lists identify prohibited and permitted taxa respectively (Simberloff 2006; Roy *et al.* 2018). While list-based approaches can be relatively straightforward to apply but fail for many cryptic species where immature individuals can be difficult to distinguish and so are frequently misidentified (e.g., aquarium fish species, juvenile crayfish) (Morais and Reichard 2018). Post-border controls often commence with compliance or quarantine inspections, which, while valuable, do not detect subclinical diseases and cannot deal with risks from newly emerging pathogens (Peeler *et al.* 2011).

Horizon scanning

Horizon scanning is the systematic process of conducting a contextualised search for potential threats and opportunities that need identification to inform future decision-making and policy development (Roy *et al.* 2014; Roy *et al.* 2019; Vilizzi *et al.* 2021). This is an essential tool for anticipating which non-native species are most likely to arrive in the scanned area and which will cause the greatest impacts, such that preventative actions can be taken (e.g., Roy *et al.* 2014; Roy *et al.* 2019). Several approaches with different strengths and weaknesses can be adopted for horizon scanning (from interview to modelling approaches; Roy et al., 2019), but generally a large set of species is reduced to a prioritized list according to the probability of their introduction, establishment, spread, and impact (although with assessments that are not as thorough as for full risk assessment). The approach on consensus-building proposed by Roy et al. (2014) for the UK has been increasingly used at national and continental levels (e.g., Peyton *et al.* 2019; Lucy *et al.* 2020). For example, Roy et al. (2019) identified 66 high-risk species at European level, with many of these then considered for full risk assessment and included in the Union list of invasive species of the EU Regulation 1143/2014 (e.g., among aquatic species (*Channa argus*, *Faxonius rusticus*, *Limnoperna fortunei*, *Morone americana*).

Risk assessment

Risk assessment is a systematic approach to prioritise both current and future threats, which assesses the scale and likelihood of arrival (through pathway analysis), establishment, spread, and impact of potentially invasive species that can be either absent or present in the assessed area. A complete risk assessment considers all of the main factors responsible for biological invasions, where Roy *et al.* (2018) identified 14 criteria for assessment, including introduction pathways, impacts on biodiversity, ecosystems, ecosystem services socio-economics, and uncertainty levels in responses. Risk assessments thus generate outputs (e.g. risk-based scores) that are suitable for policy development on invasive species, and managing invasions impacts (Roy *et al.* 2018; Robertson *et al.* 2021). They can also be used developing draft lists of species (e.g., black (prohibited) or white (permitted)), and are important components of trade rules relating to importations (e.g., a risk assessment must have been conducted that identifies significant risk of establishment and harm before a species can be banned from importation) (Robertson *et al.* 2021). Risk assessment is widely used around the world and has produced many lists of prioritized species at national, regional, and continental levels (e.g., Kolar and Lodge 2002; Peyton *et al*. 2019). Examples of risk assessments for a wide range of freshwater species (and other taxa) are available for a number of regions, including England (GB NNSS 2022a), Australia (e.g., Queensland; Queensland Government 2021) and the European Union (CIRCABC 2022).

Different forms and tools of risk assessment exist (Roy *et al.* 2018), with qualitative or quantitative outcomes. The set of minimum risk assessment standards proposed by Roy et al. (2018) included uncertainty and impacts on ecosystem services, as well as assessing invasiveness both under current and future climate conditions. Risk assessment (coupled with initial horizon scanning to prioritise species for full assessment) is thus advantageous for competent agencies to implement as resources can be focussed on the most likely high impact invaders, with appropriate targeted management actions. It is important to couple any risk assessment with the appropriate risk management when considering the feasibility and costs of the species that are then prioritized for management (Robertson *et al.* 2021).

Biosecurity and education

'Biosecurity' involves measures taken to reduce the risk of accidental introduction and spread of invasive species. National biosecurity campaigns, such as 'Check Clean Dry' in the UK and 'Clean Boats Clean Waters' in the USA, aim to raise awareness of non-native species and biosecurity, and provide clear guidance to stakeholders to reduce the risk of the spread of these species (GB NNSS 2022b). These campaigns focus on three simple steps - visual inspection, cleaning, and drying - to remove and/or kill non-native species that are attached to vessels. Specific practices, including the use of hot water and duration of drying, have been informed by studies assessing mortality of freshwater non-native species (e.g., Shannon *et al.* 2018; Bradbeer *et al.* 2020). In recent years, further specific biosecurity guidance has been developed for high risk pathways, such as angling and boating, although the efficacy of such schemes can depend on the availability of appropriate cleaning facilities (Sutcliffe *et al.* 2018). Engagement by sporting national governing bodies, such as the Angling Trust and British Canoeing in the UK, has enhanced both the appropriateness of guidance and the distribution of education material to members.

Assessing water users' compliance with biosecurity guidance presents challenges (Golebie *et al.* 2021). Since the campaign launch in 2011, awareness of 'Check Clean Dry' in the UK has increased amongst water-users, as has the number of anglers and boaters reporting compliance with biosecurity behaviours, although the risk of non-native species spread via these pathways remains apparent (Smith *et al.* 2020). The identification and engagement with all pathways are vital and there is growing recognition of the risk presented by, and biosecurity requirements of, field-based operations in governmental, private, and educational sectors (Sutcliffe *et al.* 2018). Whilst education is an important pillar of biosecurity interventions, campaigns must enhance stakeholders' motivation, capacity, and opportunity to comply with biosecurity guidance (McLeod *et al.* 2015).

3.2 Preventing invasions of newly introduced species

Early detection of newly introduced species is key for preventing their establishment (Vander Zanden and Olden 2008); this requires methods to reliably detect species when they are at low abundance levels (Britton et al. 2011c). Although this is often difficult with capture-based methods, environmental DNA (eDNA) increasingly provides a rapid and low-cost alternative (Larson *et al.* 2020), where 'first detection' capability can be targeted around high-risk introduction areas (e.g., urban centres, ports). Previously time-consuming sample filtering and contamination issues are being addressed with automated sampling equipment, with real-time methodologies also available (Doi *et al.* 2020). Metabarcoding approaches with eDNA can screen for multiple species simultaneously and could be used as a standard screening technique for non-native species in freshwaters (King *et al.* 2022).

Other early detection methods include harnessing or interrogating citizen science platforms across biomes or interrogating social media and internet sources, such as iEcology approaches (Jarić *et al.* 2020a; Unger *et al*. 2021) and aquatic culturomics (Jarić *et al.* 2020b). Citizen science is already contributing to formal records of alien species in marine ecosystems (Kousteni *et al*. 2022). Harnessing the large numbers of naturalists/general public, a high risk group that recreate on or around aquatic habitats, will greatly enhance detection capacity and speed, with many platforms now collecting high resolution georeferenced images from smartphones, with a series of experts quickly verifying images to species (Larson *et al.* 2020).

Preparation of rapid response plans for priority taxa can enhance the likelihood of eradicating or controlling newly detected species or populations. Consideration of which taxa to plan for could include the development of national 'watch' lists (Reaser, Frey and Meyers 2020a). Contingency plans for rapid response can include the most effective means for delineating the extent of dispersal, focussed effective control methods, responsibilities, resources, partners to be involved, and monitoring requirements, with identification of the lines of authority and funding also being critical components (Beric and MacIsaac 2015; Reaser *et al.* 2020b; Section 4.1). However, early detection can be hampered by incomplete taxonomy and misidentifications (Ng *et al.* 2018; Palandačić, Witman and Spikmans 2022).

Eradication of many freshwater invaders is possible, but generally at small scales and is costly, usually requiring lengthy management effort (Rytwinski *et al.* 2019). However, successful eradications have occurred for crayfish (Peay *et al.* 2006; Kouba, Petrusek and Kozák 2014; Duggan and Collier 2018), fish (Bardal 2019; Rytwinski *et al.* 2019), plants

(Simberloff 2021), molluscs, and other freshwater groups (Roda *et al.* 2016; Hammond and Ferris 2019). Methods include mechanical removal (for fish and crayfish: nets, traps, harvest, electrofishing; for plants: hand-pulling, cutting, dredging/drying), biocides or herbicides (rotenone, antimycin (fish), synthetic or natural pyrethroid (crayfish)), selected herbicides (plants), and biocontrol (diseases, predators/consumers (plants and fish)), and habitat manipulation (lake/wetland drawdown, draining; flow management, physical habitat alteration) (Hussner *et al.* 2017; Rytwinski *et al.* 2019; Sandodden 2019; Simberloff 2021). Chemical treatments are probably the most effective for eradications of fish and crayfish (Rytwinski *et al.* 2019; Sandodden 2019) but have non-target effects and, consequently, significant policy and legislative hurdles.

3.3 Eradicating and controlling dispersing populations of invaders

The reason why early detection and rapid response measures are important to implement is that once a non-native species invades a relatively large area, its eradication becomes highly challenging and expensive. Where risk assessment has indicated a specific invader is of high risk of impacting freshwater biodiversity, but surveys have indicated it has already achieved a wide spatial distribution, then decisions on the most appropriate management actions must also incorporate feasibility assessments and resourcing (Britton *et al.* 2011b). In situations where resources are limited, they are likely to be applied more effectively against a newly introduced or recently established species, rather than one that has already dispersed widely (Britton *et al.* 2011b). Indeed, the success of an eradication effort is dependent on a myriad of factors ranging from the species' invasion biology and distribution, the recipient ecosystem (habitat, connectivity etc) and the available control measures.

Where population eradications are assessed as the commensurate management action against an invader then the eradication method must be selected. This method selection should aim to consider all possible foreseen risks associated with the method, especially where the method is not species selective, such as the applications of chemical treatments against invasive crayfish and fishes (Simberloff 2009). Eradication has been an effective management tool in removing terrestrial invaders from islands, including rodents, plants, and insects (Howald *et al.* 2007; Simberloff 2009) - and river basins effectively represent 'biogeographic islands' that can provide a closed management area (Leprieur *et al.* 2009; Saunders *et al.* 2010). Thus, using river basins as management units for freshwater non-native species can provide discrete spatial areas in which eradication and control programs can be implemented that considers both the native biodiversity present (e.g., the extent of endemism) and extent of extant invasions.

The spatial extent of the invasion can mean that an eradication attempt is not feasible. Alternatively, an eradication attempt might have been attempted but was unsuccessful, such as attempts to prevent the establishment and invasion of zebra mussel in Lake Winnipeg through potash application (Depew *et al*. 2021). In both cases, population control and containment methods can then be considered, where the ultimate aim is to reduce invader impact (control) and stop their further spread (containment) (Fig. 1; Table 1; Britton *et al*., 2011a,b; Rytwinski *et al.*, 2019). Indeed, the concept of functional eradication has recently been proposed by Green and Grosholtz (2021), where they suggest that in situations where absolute eradication is not feasible, suppressing invasive populations to a level where their ecological impacts are minimised within high priority habitats is more appropriate. To reduce invader abundance to levels where their impacts are reduced or eliminated, the influence of control methods on the magnitude and direction of invader impacts must be understood. This can be complex, as impact severity can also be influenced by the life-stages present and time since introduction (Vilizzi, Tarkan and Copp 2015; Haubrock *et al.* 2022), plus a range of mechanistic context dependencies (i.e. impacts contrast under different ecological and spatiotemporal conditions) (Catford *et al.* 2022). Moreover, the relationship between invader abundance and impact is often non-linear, where impact accelerates considerably once abundance thresholds are exceeded (Jackson, Ruiz-Navarro and Britton 2015; Vilizzi, Tarkan and Copp 2015). In these situations, control methods would need to reduce abundances below these thresholds to minimise impact. Containment measures that restrict the invader to its current range include screening and barrier construction that prevent dispersal; these are increasingly applied at local or small scales to protect threatened taxa or other high-value assets (e.g., wetlands) (Dunham *et al.* 2020; Jones *et al.* 2021). Eradication and control methods that have been used against a range of invasive taxa in freshwaters are now discussed using a case study approach, where emphasis is both on the effectiveness of the control measure(s) and the response of native communities to resultant reductions in invader abundance.

Case study 1: Eradicating non-native fishes in South Africa

Within South Africa's Cape Floristic Region, non-native fishes such as smallmouth bass *Micropterus dolomieu*, bluegill *Lepomis macrochirus*, sharptooth catfish *Clarias gariepinus*, rainbow trout *Oncorhynchus mykiss* and banded tilapia *Tilapia sparrmanii* have impacted more than half of the 42 fish species native to this area (Weyl *et al.* 2014; Ellender *et al.* 2017). The Rondegat River reach, a system invaded by *M. dolomieu*, underwent South Africa's first non-native fish eradication and river restoration in 2012 and 2013. Using rotenone, a fish killing pesticide, the eradication project was conducted along a 4 km stretch of the river with the ultimate goal of protecting and preserving local fish species. Within the invaded lower reaches, native *Labeobarbus capensis* was detected at very low densities, while three other native fish species (i.e., *Austroglanis gilli*, *Barbus calidus*, *Pseudobarbus phlegethon*) were not detected but they were observed within the non-invaded zone (Weyl *et al.* 2014). Furthermore, non-native fishes were not detected above a barrier waterfall that was 5 km upstream of the river's confluence with a reservoir. Invertebrate assemblages were found to be sensitive to invasion from *M. dolomieu*, with the non-native fish eliminating native insectivorous fish predators, thereby reducing predation on invertebrate prey, with consequent food web effects (Lowe *et al.* 2008). In the presence of *M. dolomieu*, Baetidae and Chironomidae abundances increased, whereas Elmidae and Heptageniidae abundances were moderately reduced (Bellingan *et al.* 2015).

A total of 470 *M. dolomieu* and 139 *L. capensis* were removed from the treatment zone during the rotenone operation, with no fish being detected within this zone after rotenone treatment (Weyl *et al.* 2013). Native fishes rapidly re-colonised this reach where the non-native *M. dolomieu* had been eradicated, with native fish densities reaching control site densities after three years of non-native fish absence. The successful removal of non-native *M. dolomieu* resulted in increased invertebrate and fish biodiversity, and this has encouraged more native fish restoration projects in South Africa, such as the Cape Floristic Region's non-native fish eradication programs within farm reservoirs (e.g., targeting *L. macrochirus*, *C. carpio* and *O. mykiss*; see Dalu *et al.* 2020). These studies provide important background knowledge for conservation authorities considering the removal of invasive non-native fishes from lotic water systems using rotenone.

Case study 2: Eradicating *Gyrodactylus salaris* from Norwegian salmon rivers

The salmon fluke *Gyrodactylus salaris* is a freshwater Atlantic salmon *Salmo salar* ectoparasite native to the Baltic region, and an invasive species in Norway that was first detected in the 1970s. This parasite is one of the most severe threats against Norwegian Atlantic salmon. It was unintentionally introduced with live fish transports and distributed in Norwegian rivers through stocking. Norwegian Atlantic salmon populations are highly susceptible to the parasite, with an average reduction of parr densities of 86 % (48-99 %) in infected rivers (Johnsen and Jensen 1991).

The Norwegian government declared a goal to eradicate the parasite from Norwegian rivers. As *G. salaris* is a viviparous, obligate parasite, restricted to freshwater, and survives for only a few days without its host, then eradication of all hosts with the chemical rotenone has been the main strategy for its eradication (Sandodden *et al.* 2018; Adolfsen, Bardal and Aune 2021). The eradication campaign has been ongoing for more than 40 years, with the parasite now eradicated from 39 of 51 Norwegian rivers where it has been detected, with four rivers still under post-treatment surveillance awaiting eradication confirmation. In areas targeted for *G. salaris* eradication, local strains of sea trout *Salmo trutta* and Atlantic salmon in the treated anadromous zone undergo an extensive program for preservation and restocking to minimise the long-term impacts of eradication (O'Reilly and Doyle 2007). Studies on benthic invertebrates in rotenone treated rivers suggest short term impacts, with increasing effects at higher water temperatures and rotenone concentrations, and longer exposure times. Although most invertebrate taxa recolonize within a year (Kjærstad *et al.* 2021), recolonization can take several years if the rotenone treatment comprised most of the basin (Mangum and Madrigal 1999).

Case study 3: Controlling invasive macrophytes

Invasive macrophyte infestations threaten freshwater ecosystems throughout the world. While a number of key traits are exhibited by the majority of invasive macrophytes which increase their invasiveness, their presence in a system is usually a symptom of anthropogenic spread combined with increasing urbanisation, industry, and agriculture, which ultimately results in eutrophication (Coetzee and Hill 2012). Both floating aquatic macrophytes, such as water hyacinth *Pontederia crassipes*, giant salvinia *Salvinia molesta*, and water lettuce *Pistia stratiotes*; and submerged invasive macrophytes such as *Elodea* spp., *Hydrilla verticillata*, *Egeria densa*, and *Lagarosiphon major*, are among the most prolific invasive aquatic weeds worldwide, with significant socioeconomic and ecosystem impacts (Cuthbert *et al.* 2021).

A number of management options are available for the control of invasive macrophytes, with varied success. Floating macrophytes may be manually removed, chemically controlled, and/or utilised, particularly by poor rural communities which are perceived to benefit from using them (e.g., as fuel in South Africa) (Hill *et al.* 2020). Unfortunately, these methods are rarely effective in the long term due to the effort and cost required to remove/control significant amounts of high water content biomass, and may even promote their spread. Biological control, using host specific natural consumers ('enemies'), has been particularly effective in controlling floating macrophyte invasions (Coetzee *et al.* 2021), especially in tropical and subtropical parts of the world. In southern Africa, the invasive red water fern *Azolla filiculoides* from South America was successfully controlled through the release of the frond-feeding weevil *Stenopelmus rufinasus*, imported from Florida (McConnachie, Hill and Byrne 2004). Their release resulted in the local extinction of red water fern at 81 % of release sites over an average of 7 months. In temperate areas, an integrated strategy using a variety of methods is often required to obtain acceptable control (Shearer and Nelson 2002).

In contrast to floating macrophyte species (whose presence is known at the early stages of invasion), submerged plant invasions often remain undetected for long periods of time, (Hussner *et al.* 2017) although that is being overcome through eDNA methods (Doi *et al.* 2021). Similar management strategies (e.g., manual removal, chemical control, shading; Schooler 2008) are also used for their control, again with varied success due to fragmentation and regeneration following control operations. While biological control of floating aquatic plants has many successful examples, the biological control of submerged aquatic macrophytes has been variable with, for example, the successful use of grass carp *Ctenopharyngodon idella* in control programs, such as in North America, being tempered with the invasion of these fish following biosecurity lapses, other than where sterile fish have been used (Chilton and Muoneke 1992).

Case study 4: Managing populations of invasive bivalves

Through the formation of dense and expansive populations, invasive freshwater bivalves such as *Corbicula* clams and *Dreissena* mussels can substantially alter ecosystem functioning and biodiversity. Notably, their presence has transformed nutrient cycling in the Laurentian Great Lakes (Li *et al.* 2021) and underpinned the escalated growth of problematic macrophytes, such as in Lough Erne, Ireland (Crane *et al.* 2020). Mass die-offs can lead to deoxygenation and acute nutrient-based toxicity, while the persistence of empty shells may detrimentally alter habitats and community composition (McDowell and Sousa 2019; Coughlan *et al.* 2022). Further, living and dead biomass can foul anthropogenic infrastructure, such as the internal surfaces of pipework and irrigation systems. Consequently, bivalve infestations along with costly management interventions often result in negative socio-economic effects (Haubrock *et al.* 2022).

The control of established populations of invasive bivalves can be exceedingly difficult, particularly in open waterbodies (Sousa *et al.* 2014), which has been evidenced by numerous attempts to control bivalves at locations across Europe and North America, such as the River Barrow, Ireland (Sheehan *et al.* 2014) and Lake Tahoe in California, USA (Wittmann *et al.* 2012). Although no evidence for complete eradication of *Corbicula* from natural waterways exists, techniques such as dredging, hand or suction harvesting, deoxygenation, thermal shock, and the application of various molluscicides can reduce *Corbicula* abundances (Sheehan *et al.* 2014; Sousa *et al.* 2014; Coughlan *et al.* 2021). Contrastingly, many of these techniques have been used to successfully eradicate *Dreissena* mussels (e.g., molluscicides, deoxygenation), such as from Millbrook Quarry, Virginia, USA (Sousa *et al.* 2014), although approaches in

Lake Winnipeg were unsuccessful (Depew *et al*. 2021). In industrial settings, physical removal, electrocution, desiccation, thermal, and chemical treatments have been successfully used to control and eradicate bivalve infestations (Sousa *et al.* 2014). Nevertheless, the efficacy of these approaches can vary and all management interventions, even those performed in industrial settings, can have substantial negative consequences for native freshwater biodiversity.

3.4 Living with invasive species

In freshwater systems with high levels of socio-economic and recreational activity, there tends to be high introduction rates of non-native species; as some of these species develop highly invasive populations, efforts to eradicate, contain and control these invasions are highly challenging (Fig. 1; Table 1). In regions where non-native species are continuing to be introduced and management efforts to control their invasions are increasingly expensive and/ or ineffective, then there is an option to also 'live with invasive species', where some populations continue to be exploited without any further control, whereas other populations might continue to be controlled and contained to prevent further damage to biodiversity and/ or ecosystem services (Fig. 1). The following case studies on the Laurentian Great Lakes, USA/Canada and Lake Naivasha, Kenya, highlight two regions where aspects of living with invasive species have been adopted.

Case study 5: Laurentian Great Lakes, USA and Canada

The Laurentian Great Lakes of North America are an interconnected system of five freshwater lakes that all rank among the seventeen largest lakes in the world (Herdendorf 1982). A multijurisdictional system, managed by two federal governments (USA, Canada), as well as multiple state, provincial, and tribal agencies, the Laurentian Great Lakes have a long history of living with invasive species (Campbell and Mandrak 2019). Records of unintentional introductions date to as early as 1819 and deliberate introduction and stocking of non-indigenous species can be found as early as 1870 (Emery 1985). Currently, the main pathways for introduction include commercial shipping, live trade, recreational boating and angling, and stocking (Mandrak and Cudmore 2010). Over 180 species that are non-native to the basin have been successfully introduced and established within the past two centuries (Pagnucco *et al.* 2015), with a small number of species having radically altered the ecosystem community structure. For example, invasion of sea lamprey *Petromyzon marinus*, along with overfishing and habitat alteration, caused devastating impacts for the Great Lakes commercial fisheries - particularly lake trout *Salvelinus namaycush*, lake whitefish *Coregonus clupeaformis*, and deepwater ciscos *Coregonus* spp. (Siefkes *et al.* 2013). Zebra and quagga mussels have also drastically altered the lakes' aquatic food webs and water clarity (Mayfield *et al.* 2021).

The aggregate cost of aquatic invasive species to the basin is estimated to be well over US$100 million annually (Fantle-Lepczyk *et al.* 2022). Interventions used against these species range from active support for maintaining populations of desirable species to multi-pronged control and eradication attempts, and the most common response, no intervention at all. The efforts to control sea lamprey are the most extensive in the basin, with a control program involving lampricide applications in combination with physical and electromechanical barriers (Siefkes *et al.* 2013).

The spatial scale of the Great Lakes, coupled with their highly modified state, means that eradication of many invaders is not a viable option. At this stage, most management efforts are vested towards prevention of new invasions, such as the extensive efforts to exclude bighead carp *Hypothalmichthys nobilis* and silver carp *Hypothalmichthys molitrix* (Kokotovich and Andow 2017) or reducing, but not eliminating, impacts from established species, such as the goals of the sea lamprey control program (Siefkes *et al.* 2013). Acceptance of the persistence of invasive species has induced the Great Lakes management agencies to optimize best suited strategies for living with these species.

Case-study 6: Lake Naivasha, Kenya

Lake Naivasha is one of only two freshwater lakes wholly in Kenya and thus provides a wide range of ecosystem provisioning services at national (e.g., agricultural/ horticultural jobs, export income), regional (e.g., potable water, fisheries, power) and international (e.g., provision of non-seasonal flowers and vegetables) scales (Hickley *et al.* 2004; Hickley *et al.* 2015). Allied to these provisioning services is the introduction of numerous non-native species in the last 100 years, including mammals, crayfish, plants and fish (Gherardi *et al.* 2011b).

The red swamp crayfish *Procambarus clarkii*, introduced intentionally in 1970 for fishery enhancement, was key in the lake's transformation from a macrophyte dominated and clear water state to an algal dominated and turbid state (Gherardi *et al.* 2011b). Despite this shift in the lake's stable state, the artisanal fishery remained dominated by two tilapia species (introduced in the 1950s) up to the early 2000s, when a combination of further lake degradation (e.g., continued nutrient enrichment) and high exploitation resulted in their catches declining until the stocks were not viable for fishing in 2009/10 (Oyugi *et al.* 2011). However, the accidental introduction of non-native *C. carpio* into the lake in the late 1990s, and their subsequent establishment, has since provided an alternative target species that is also capable of tolerating the increasingly eutrophic and degraded lake conditions (Hickley *et al.* 2015).

Although *C. carpio* catches supported the fishery catches throughout the last 20 years, local fishers and consumers continued to prefer tilapia and so the fishery management of the lake decided to release the non-indigenous Nile tilapia *Oreochromis niloticus* in 2011, a globally invasive species that can also tolerate relatively degraded conditions, with *Clarias gariepinus* also escaping into the lake from on-shore aquaculture systems (Hickley *et al.* 2015). Both species established and now feature in fish catches (Waithaka *et al.* 2020; Keyombe, Waithaka and Obegi 2015). Thus, to maintain and enhance food security in the region, fishery managers have manipulated the fish community of the lake using non-native fishes, with these species providing abundant stocks for exploitation, despite the shift in lake stable state and on-going nutrient enrichment. Active management of these fishes primarily relates to the enforcement of fishery regulations that determine the extent of fishing pressure and monitors catch rates by species. Legislation is in place that means new introductions can be completed after risk assessment, although issues of food security and the potential for deriving socio-economic benefits from new species mean that these introductions of ecologically damaging species might still occur (Hickley *et al*. 2015).

Table 1. Summary of the three sequential stages, and their actions and details, that can be brought together within strategies to bend the curve of freshwater biodiversity loss from the harmful effects of freshwater non-native species.

Stage	**Action**	**Detail**
1. Introduction prevention	Legislation and regulation	Only permitted/ approved non-native species can be imported/ introduced/ used in closed aquaculture, and following full risk assessment
	Horizon scanning	Identify high-risk species most probable to be introduced in near future
	Pathway surveillance	Implement surveillance of introduction pathways (e.g., for species from horizon scanning)
	Import inspections/ quarantine	Qualified personnel inspect imports for non-native species presence/ use quarantine to provide time for detection
	Enforcement of legislation/ species' lists	Enforce legislation via regulatory frameworks, including lists of permitted/prohibited species (i.e., risk-based)
	Biosecurity approaches	Provide infrastructure and mechanisms for freshwater users to decontaminate equipment of possible non-native species
2. Preventing an invasion following an introduction	Early detection of new introduction	Methods needed to detect newly introduced species prior to establishment, especially in high-risk areas (e.g., near ports, urban centres etc.), including predictive tools.
	Rapid response decision	Following detection, risk-based decisions needed on the appropriate response to protect biodiversity
	Implementation of the rapid response	Where eradication is the decision, rapid implementation is needed to remove introduced individuals before establishment

3. Managing extant invaders	Risk-based management decisions	Where the invasion is underway, risk-based decisions are needed on how to protect/ restore biodiversity from harm
	Control invader abundance	Where local biodiversity impacts are a function of invader abundance, removals can reduce population sizes, including to zero by eradication and suppression to levels where ecological impacts are minimised ('functional eradication'). Alternative control methods seek to reduce invader abundance through impacting recruitment success (e.g. sterile male release techniques).
	Contain invader to current range	Management actions are implemented that prevent the further dispersal of the invader through connected waters and through anthropogenic means
	Accept invader presence (including monitoring their populations and increasing resilience to impact of native communities)	Where the invader has wide spatial distribution, low risk to biodiversity and/ or control methods are ineffective, an active decision to accept invader presence/ do-nothing is acceptable. Can be coupled with management of physical environment to enhance populations of native species, including measures to increase recruitment and competitive abilities, as well as continued monitoring of the effects of invasive populations on biodiversity, ecosystem functioning and services, with instigation of management and conservation actions should unacceptable impacts start to be detected.
Applicable to all stages	Risk assessment	Process to assess risk posed by non-native species in the environment (assessing probabilities of entry/ establishment/ dispersal/ impact)
	Biosecurity of sites	Sites containing non-native species (aquaculture, fisheries, ornamental) should be bio-secure to minimise colonisation pressure

Education programs	Programs to educate policy makers, practitioners, stakeholders and the general public are needed for all stages and actions. Could be completed with social assessments to inform reasons for non-compliance with regulations and/or management programs.

4. Implementation

4.1 Implementation strategies and contingency plans

The approaches and measures for managing freshwater non-native species in Table 1 can be incorporated into species-specific strategies through the step-wise process outlined in Figure 1. However, when management is dealing with multiple non-native species, where concomitant actions range from introduction prevention to minimising invasives' impacts, these processes can be integrated into wider strategies. Generic contingency plans can outline the actions needed on the first detection of a non-native species (Raymond *et al.* 2011), such as that developed in England for actioning when a disease not native to the European Union was detected, which comprised of a framework response plan to assist and direct the Government response and an operation manual to manage the 'on the ground' response (Oidtmann *et al.* 2011).

For very high-risk species that have yet to be introduced, species-specific contingency plans can be formulated that are implemented on its first detection in that region (Raymond *et al.* 2011). For example, for the countries of the UK, a specific contingency plan has been developed for *G. salaris* (Oidtmann *et al.* 2011), given the high risk it poses to wild and captive populations of salmonid fishes (Case Study 2, Section 3). The contingency plan for Scotland outlines the legislation that permits the control of the parasite, along with the management details, including the roles and responsibilities of government departments and agencies, a risk assessment, details on decision making for containment or eradication, the eradication methods and their application, gene banking processes, and resources (Scottish Government 2011). This ensures that decisions on how to manage the species are made in advance, which can be important given they involve the resolution of ecological, socio-economic, and ethical arguments among conflicting groups (Stokes *et al.* 2006; Finnoff *et al.* 2007). Similar contingency planning exists for other harmful invasive species, such as the Invasive Mussel

Collaborative, which comprises 36 different federal, state, provincial, and tribal agencies, plus non-government groups, research institutions, private industries etc., who work together to address the risk of, and response to, zebra and quagga mussel invasions in the US (https://invasivemusselcollaborative.net).

4.2 Implementation challenges and impediments to managing non-native freshwater species

Although some of the challenges in managing non-native species have been overcome (e.g., substantial developments in invasion risk assessments; Section 3), many still remain. For example, in some jurisdictions, there is no legislation for managing non-native species in freshwaters and/ or there is legislation that promotes their use in aquaculture. In many freshwater biodiversity-rich nations such as Brazil and India, legislation permits the use of non-native fishes within freshwater aquaculture (e.g., Brito *et al.* 2018; Singh 2021), with these species then escaping into the wild during biosecurity lapses (Casimiro *et al.* 2018) and/ or extreme climatic events (Raj *et al.* 2021). Even where national legislation does exist to prevent introductions, the introduction pathway of many freshwater non-native species involves global trade routes and importing species through busy traffic areas (e.g., ports, airports). The opportunities for detecting contaminants of legal animal movements or the illegal import of species is limited (Chapman *et al.* 2017). Nevertheless, the success of ballast-water screening at reducing the numbers of introduced species into the North American Great Lakes (Ricciardi and MacIsaac 2022) emphasises that where pathways are targeted with effective and enforced legislation, strong outcomes for the protection of freshwater biodiversity are possible.

The growth of trade through routes that are largely unregulated (e.g., the internet) has also enabled non-native species to be traded in a manner that is difficult for authorities to track (Olden, Whattam and Wood 2021). The potential for unregulated internet trade to provide an introduction pathway for fishes was revealed in Brazil, where a six-month monitoring period of social media groups recorded over 1100 posts advertising the sale of over 5000 specimens of over 600 species, of which 66 % were non-native and 25 % were forbidden by national legislation to be traded (Hirsch *et al.* 2021). There was also a trend for more expensive prices to be associated with species that were non-native, prohibited, and of larger body size (Hirsch *et al.* 2021). Similarly, well-intended religious practices (e.g., Tsethar in Buddhism) have resulted in the release of significant numbers of invasive species, including African catfishes and non-native turtles (Everard *et al.* 2019). The complexities involved in the regulation and enforcement of religious and cultural practices associated with live release means that community-supported and voluntary legislation need to be developed to ensure rituals do not result in environmentally harmful outcomes (Everard *et al.* 2019).

These issues of unregulated trade are compounded by a general lack of investment in introduction prevention schemes in many world regions, despite the ability of these schemes to provide substantial long-term savings (Ahmed *et al.* 2022). Counter-intuitively, preventative management is sometimes viewed as riskier than control, as its effectiveness (or lack thereof) at preventing invasions cannot be easily predicted (Finnoff *et al.* 2007). Where budgets are limited but there are multiple conservation demands, expenditure on non-native species is often delayed until impacts have been demonstrated (Ahmed *et al.* 2022). Indeed, Finnoff et al. (2007) suggested that such paradoxical decisions stem from the association between preferences for risk-bearing and the technology of risk reduction, with risk-averse managers more likely to use invasion control than introduction prevention, as control can appear less risky (i.e., invaders are seen being removed). Effective management of non-native species also requires considerable political acumen and commitment to liaise with multiple agencies and stakeholder groups to agree on actions. Risk-aversion, coupled with lacking commitment to coordinating and implementing preventative actions, might thus result in river basins with high invasion rates.

A major implementation challenge to protect and restore fresh waters from non-native species is that these species might represent only one of numerous, potentially interacting, stressors in the environment (Craig *et al.* 2017). Species rich communities are generally considered as providing substantial resistance to the establishment and invasion of introduced species (Alofs and Jackson 2014). However, biotic resistance tends to be relatively weak in fresh waters that have been disturbed through other anthropogenic activities, such as through increased resource availability (e.g., through nutrient enrichment) and modified physical structure of the ecosystem (e.g., river impoundment) (Catford, Jansson and Nilsson 2009). Riverine impoundment generally leads to species-shifts from lotic to lentic and specialist to generalist (Noble *et al.* 2007), which often favour non-native over native species, and potentially elevates propagule pressure through increased recreational use (Johnson, Olden and Vander Zanden 2008). For example, the creation of multiple reservoirs by hydroelectric dams in Southern Brazil has provided opportunities to create sport fisheries based on non-native species such as peacock basses (*Cichla* spp.) (Espínola, Minte-Vera and Júlio 2010), where high predation rates from their invasive populations further decrease native fish species richness and abundance (Britton and Orsi 2012; Pelicice and Agostinho 2009). Indeed, evidence from the USA revealed the likelihood of finding non-native species in impounded rivers was up to 300 times higher than in natural lakes, with reservoirs frequently supporting multiple invaders (Johnson, Olden and Vander Zanden 2008). Correspondingly, non-native species management strategies should be implemented in tandem with other strategies within the wider freshwater biodiversity restoration toolbox to ensure holistic approaches are implemented.

Although methods to remove non-native species from fresh waters exist, their application might not always be considered feasible (e.g., the method is ineffective over large spatial areas) or ethical (e.g., due to high mortality rates of non-target species) (Britton *et al.* 2011b). For example, although toxin application is effective at killing target species, it is also usually non-species specific, often resulting in high mortality rates in other species (Britton *et al.*, 2011a), and therefore impractical in river systems harbouring high numbers of both endemic and non-native species. High losses to non-target species have been seen in invertebrate communities in ponds and lakes treated with the piscicide rotenone, although these populations generally recover rapidly (e.g., within six months; Britton and Brazier 2006; Dalu *et al.* 2020). Moreover, even where eradication attempts are made, failures do occur, such as the failed attempt to eradicate Northern pike *Esox lucius* from Lake Davis, USA (Lee 2001). However, these unsuccessful management operations can be used to inform future attempts (Rytwinski *et al.* 2019).

Novel control measures based on exploiting innate behaviours or weaknesses (‘achilles heel’) of invaders are increasingly gaining attention. For example, the exploitation of jumping and pushing traits are being utilised in trap, screen, and weir designs (Holthe *et al.* 2005; Stuart and Conallin 2018; Tempero *et al.* 2019) and the overwintering or spawning aggregations/locations of invaders are now used to focus removal actions, such as *C. carpio* in Australia (Taylor *et al.* 2012; Sorensen and Bajer 2020). Moreover, genetic techniques for invasive fish and crayfish control continue to be explored (e.g., daughterless *C. carpio* in Australia; Thresher *et al.* 2014); sterile male release (e.g., non-native crayfish in England; Green *et al.* 2022), as are gene drives (Bajer *et al.* 2019) but have not yet been implemented on a large scale (Kopf *et al.* 2017; Simberloff 2021). Moreover, the use of ‘ark’ sites, where imperilled native species are protected from extinction through their translocation from donor sites (e.g., threatened with invasion by high impacting species) to refuge sites (e.g., of low invasion risk), can provide extra time for more effective control methods to be developed and then deployed on the high-impacting invader, as seen with imperilled populations of White-clawed crayfish *Austropotamobius pallipses* in the British Isles that are threatened by invasive North American crayfishes (Nightingale *et al.* 2017).

Even where risk assessment processes have identified a species as being of high risk to freshwater biodiversity, it does not necessarily follow that management actions commensurate with that risk level will follow. For example, analysis on the global application of the risk assessment tool 'Freshwater Fish Invasiveness Screening Kit' (FISK) revealed *C. carpio* was the most widely screened species, with it assessed as having a high risk of invasiveness in all regions (Vilizzi *et al.* 2019), with the species already being recognised as high impacting on most aspects of freshwater biodiversity and functioning (Vilizzi, Tarkan and Copp 2015). However, in countries such as England, the socio-economic value of the species in angling and aquaculture, coupled with its long introduction history and wide spatial distribution, means that it is managed as a naturalised species and so falls outside of legislation controlling non-native species (Hickley and Chare 2004). Similarly, the species has also been translocated widely across Turkey and now supports productive fisheries, despite all risk screenings indicating the species will become highly invasive in the wild (Vilizzi *et al*. 2012).

5. Conclusions

Bending the curve for freshwater biodiversity by protecting and restoring species from the harmful effects of non-native species remains highly challenging. Introduction rates of species continue to accelerate globally and although not all of these species will become invasive, those that establish and disperse have the potential to cause deleterious impacts on freshwater biodiversity. Notwithstanding, there have been considerable developments in the last decade on horizon scanning and risk assessment processes that provide the basis for strong introduction and invasion preventative actions, especially when coupled with effective legislation and regulation that enables surveillance of introduction pathways, and implementation of rapid detection methods and reaction protocols. Also, a range of removal methods are available to control non-native species in both lentic and lotic systems and, in some cases, eradicate high-impacting populations. Although these methods tend to be of low technology (e.g., toxins), new methods are being developed that potentially provide more species-specific approaches, with ark sites providing potential protection for highly imperilled native species. However, implementation challenges and impediments to management success remain, including the application of precautionary principles to prevent introductions often being viewed as a risky option and non-native species often acting as an additional stressor in already highly stressed and modified ecosystems. Overcoming these challenges and impediments is, however, crucial if freshwater biodiversity is to be protected and restored from the damaging impacts of non-native species invasions within the Emergency Recovery Plan and so contribute to bending the curve of freshwater biodiversity loss.

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Box 1. Definition of terms used in the paper (note these are defined in freshwater terms)

Term	Definition
Non-native / non-indigenous / alien	A species with a natal origin outside of, or foreign to, the waterbody/ river basin under discussion.
Introduction	The deliberate or accidental release of a species into a waterbody / river basin where it is not found naturally.
Introduced species	A species that has been released into a waterbody / river basin for the first time.
Pathway	A route / mechanism providing the entry of a non-native species into a waterbody / river basin (and its subsequent introduction).
Establishment	The production of a sustainable population from the introduced individuals.
Dispersal	The spread of individuals from the invasion front into areas where the species has not been found previously.
Invasive species / Invasive alien species / Invasive non-native species / Invasive non-indigenous species / Invader	A species with a natal origin outside of, or foreign to, the waterbody/ river basin under discussion that has been introduced, established, and dispersed, and is impacting native biodiversity.
Eradication	The complete removal of all life-stages of the invader from a waterbody / river basin through management actions.
Control	The intentional reduction in invader population abundance (as number and / or biomass) to levels which reduce its impact on native biodiversity.

Containment	The intentional restriction of the invader to its current distribution to prevent its spread.