FISHERIES SOCIETY OF THE BRITISH ISLES
BRIEFING PAPER 5

NON-NATIVE FRESHWATER FISH INTRODUCTIONS[1,2]

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EXECUTIVE SUMMARY

Preamble: The aim of this Briefing Paper is to provide a contemporary account of knowledge on various aspects of introductions of non-native fish species. It reviews issues associated with introduction pathways, ecological and economic impacts, risk assessments, management and climate change perspectives and offers guidance to help reconcile the need for long-term sustainability of aquatic biodiversity and the increasing demand of relevant stakeholders to diversify their activities using non-native species.

Section 1: The introduction of non-native freshwater fishes is of increasing concern to policy-makers, managers and scientists, with the number introduced worldwide doubling in the past 30 years. Growth of global trade, demand for new products and human mobility have been principally responsible for this increase and biological pollution as well as subsequent invasions are now a major environmental issues of public concern. At present, scientific understanding of the effects of non-native fish introductions is characterised by disputes over definitions, resulting in the understanding of ecological impact being blurred and messages weakened. Here, the terminology is based on the common understanding that the introduction of a non-native species is primarily an ecological issue and so should be defined as such.

Section 2: The introduction of fish species into fresh waters is commonplace around the world, with the principal reported motives for introductions being aquaculture (39%) and improvement in wild stocks (17%). Others, such as bioengineering, ornamental trade and research activities, also play a role. In all cases, aquaculture plays a major role in the provision of stocking material, but in terms of colonisation and invasion routes, a number of natural and human-mediated mechanisms are known. In Europe, there are effectively three invasion routes that are responsible for the dispersal of aquatic organisms, consisting of an interlinked network of 30 main canals with more than 100 branches and more than 350 ports.

Section 3: There are several major ecological impacts often associated with non-native fish introductions, including predation, habitat degradation, competition for resources, hybridisation and disease transmission. However, consideration of these

aspects in isolation is rarely sufficient to characterise an ecological impact of an introduced non-native species. Instead, there is a requirement for these to be associated with a quantifiable and significant decline of biological or genetic diversity threatening the long-term integrity of native species. Understanding ecological impacts is not just about determining ecological changes, as these are inevitable when any species is introduced and becomes established in a new ecosystem outside its native range. Rather, it is whether these changes lead to a measurable loss of diversity or changes in ecosystem function. Only if this happens can the introduction of that species be considered harmful.

Section 4: Economic activity, particularly globalisation through trade, is the principal driver of human-mediated non-native fish introductions. For example, approximately 17 % of the world's finfish production is based on non-native species; production of the African cichlid tilapia is much higher in Asia (>1.6 million t in 2005) than in most areas of Africa (245 000 t); introduced salmonids in Chile support a thriving aquaculture industry that is responsible for approximately 30 percent of the world's farmed salmon and directly employs approximately 30 000 people. Consequently, these economic benefits should be balanced against the detrimental environmental, social and economic impacts of introduced non-native fishes.

Section 5: To justify measures to mitigate threats to biodiversity from non-native fishes, it was apparent that protocols were needed to assess the risks of introductions. To this end, existing tools for non-native plants and plant pests were adapted to create a two-phase sequence of protocols. For the initial (hazard identification) phase, the Fish Invasive-ness Scoring Kit (FISK) was developed from a pre-existing weed risk assessment. Species identified by FISK as potentially invasive can then be subjected to an adapted form of the European Plant Protection Organisation scheme to assess the ecological, social and economic risks posed by the species, such as has the generic GB Non-Native species Risk Analysis scheme (GBNNRA) or the risk protocols recently developed for the use of alien species in aquaculture in the European Union.

Section 6: Prevention of introductions is only the starting point of the management of non-native fishes. Once an unregulated introduction of a non-native fish has occurred, then a risk-based response is required to determine the most appropriate tool(s) to

manage that species. Determination of the most appropriate management responses to deal with an established non-native fish requires use of risk assessment and risk management. Risk assessment characterises the likelihood and severity of potential adverse effects of the introduced fish whereas risk management identifies, evaluates, selects and implements the actions designed to reduce that risk. Management responses are typified by either a remediation or mitigation response: remediation aims to provide a solution through extirpating the species, such as through eradication, whereas the aim of mitigation is to minimise the impact of the species on the provision of ecosystem services. Eradication exercises on introduced non-native fishes remain rare and success is usually limited to closed water bodies, suggesting that mitigation strategies will become increasingly important as further introductions occur.

Section 7: In the northern hemisphere, the consequences of predicted temperature increases are typically described as shifts in community structure towards more thermophilic and southern taxa. Given the ecological sensitivity of some northern habitats, colonisation by non-native fishes may have serious consequences for many indigenous fish species. An extremely important implication of climate change concerns those habitats already subjected to introductions of non-native fishes that are still within a lag phase prior to the species expansion. Events that may trigger termination of a lag phase include increased temperatures that would enable successful reproduction and recruitment, and decreased biological resistance and resilience in the ecosystem arising from increased thermal stress on native species. Should native communities be modified and/or destabilised by temperature increases, invasion pathways for some introduced warm-water fishes may open further. This is especially the case for those species that have already been purposely introduced into cool-water, temperate regions (e.g. for aquaculture and angling) in the belief that they could not become invasive because of thermal barriers to reproduction. However, increased waters temperatures may also bring some socio-economic benefits, particularly in relation to aquaculture production in temperate regions through reduction in current climatic limitations thus facilitating an increase in the diversity of species being cultured.

Conclusions:

Based on current scientific knowledge (ecological impact) and the ongoing demand for non-native fish introductions (economic/societal interests), this briefing paper takes a pragmatic approach to the neo-paradox that has resulted from post mid-twentieth century economic development: *Despite society's knowledge of the potential adverse impacts on recipient ecosystems, economic drivers are still pushing forward for new introductions of non-native fishes*. Acknowledgment and understanding of this paradox is crucial to ensure successful regulation of undesirable non-native species introductions. It should enable market diversification to proceed in conjunction with robust risk assessment tools, more efficient mitigation schemes and, ultimately, the more efficient regulation of future introductions.

1 INTRODUCTION

1.1 Aims & intended audience

The aim of this Briefing Paper is to provide an account of current knowledge on various aspects of the introduction of non-native fishes[3]. It was prepared by a group of scientists in ecology, aquaculture, fisheries, policy and management, with experience in the provision of advice to national and international governmental bodies, including the European commission. It provides a comprehensive review with far reaching conclusions that aim to reconcile the prospect of economic growth with long term sustainability of aquatic biodiversity. Increasingly, the need to develop aquaculture as a substitute for commercial exploitation of wild stocks is reliant on the aquaculture sector diversifying through farming new non-native fish species, while characterising, assessing and managing the risks for aquatic ecosystems. This document aims to benefit policy makers, environmental managers, conservation groups, NGOs, the aquaculture sector, the ornamental fish trade sector and members of the general public, as well as scientists focused on the conservation and preservation of global biodiversity.

1.2 Why is the introduction of non-native freshwater fishes a concern?

The number of species introduced worldwide has more than doubled (Gozlan 2008) compared with estimates nearly three decades ago (Williamson & Fitter, 1996), with growth in global trade and human mobility principally responsible (Sala *et al.*, 2000; Gozlan, 2008). Biological invasions are now considered as a major environmental issue of public concern.

The introduction of a non-native species in an ecosystem is always likely to present an ecological risk if the species is able to integrate itself successfully into the ecosystem (see Section 2), resulting in possible detrimental impacts on native species or even on ecosystem functioning. The introduced species could impact biodiversity through predation, competition, hybridisation, habitat modification and/or transmission of a

[3] This review specifically deals with the introduction of non-native species and not the mixing of strains or genetic variants of the same species that are introduced into a new environment already inhabited by an indigenous population of the same species.

novel disease. There are many examples of such impacts, with some cases causing serious consequences for the conservation of biodiversity (see Section 3). Fishes are among the most introduced group of aquatic animal in the world (i.e. 624 species, Gozlan 2008) and also one of the most threatened, with the total number of threatened fish species reaching 1201 in 2007 (IUCN, 2008). Nonetheless, fish species are still introduced around the world because of societal demands for fish products for food aquaculture (51 %), ornamental fish (21 %), sport fishing (12 %) and fisheries (7 %) (Gozlan 2008). Whilst socio-economic forces suggest the increasing trend of non-native fish introductions will continue, it implies the associated risk of ecological impact and biodiversity loss will also increase. This is characteristic of many other ecological issues where the needs of societal development do not necessary converge with conservation interests (See section 4).

When the issue is examined more closely, however, these societal needs for non-native fish can be regulated, and not all non-native fish introductions result in ecological impact and / or loss of biodiversity. Whilst it is crucial to be better able to forecast the risk of impact resulting from a non-native fish introduction, given that risk cannot be totally absent, then it is crucial to develop appropriate management tools and mitigation protocols (see Section 5 &6). The current scientific understanding of non-native fish introductions is highly polarised, as many specific terms remain ill-defined, with a lack of common understanding of aspects such as ecological impact. At a time when policies dealing with non-native species are being put in place, it is important that the current understanding of this issue is refined and reframed in a global context of environmental pressures (see Section 7).

1.3 Definition of terms

As there are so many different terms used to describe biological invasions, it can be difficult to determine whether common thinking is being applied across different terminologies. For example, non-native fish are also termed exotic, non-indigenous, alien, xenic, noxious, weedy, pest and foreign (Peretti, 1998; Daehler, 2001; Occhipinti-Ambrogi & Galil, 2004; Copp *et al.*, 2005a). Additionally, 'invasive species' is a term that is often used when 'invasive population' would be more appropriate. Despite attempts to achieve common definitions, difficulties remain because of a combination of ecological and political perspectives being used to

determine those species that are either 'native' or 'non-native' (Shafland & Lewis, 1984; Davis & Thompson, 2000; Shrader-Frechette, 2001; Copp *et al.*, 2005a). This is evident from the common use of national boundaries to determine whether a species is native or non-native, irrespective of the biogeography of the species concerned. Often, the term translocation is used to differentiate between within- and across-country movements. In terms of ecology, this has little relevance as a species moved from one basin to another basin within the same country could generate similar ecological outcomes (i.e. predation, competition, hybridisation, habitat use, disease transmission) as a species moved across national borders. The International Union for Conservation of Nature (IUCN) and the Convention on Biological Diversity (CBD) have also called on parties to focus on non-native species at the ecosystem level, rather than the national level, as both organisations define non-native species as a species introduced outside its natural range (Riley, 2005). For these reasons, tight definitions are suggested that are based on the common understanding that species introductions are primarily an ecological issue and as such, definitions should follow an ecological perspective rather than a political one (Table 1).

However, there are some aspects that cannot be successfully defined using an ecological perspective alone (Davis *et al.*, 2001). These require strict policy guidance, for example the definition of the persistence of a species that may otherwise be considered naturalised. There is no clear ecologically relevant time limit that could be confidently used to set the time limits of when an introduced non-native species could be considered naturalised, so a subjective decision is necessary (Davis & Thompson, 2000; Falk-Petersen *et al.*, 2006). This is an important consideration, because a species considered as persistent and naturalised may be managed differently to a species that has been introduced more recently.

Table 1: Terms and definition based on an ecological approach to fish introduction[4].

Term	Definition
Native	A species that occurs naturally in a specific water (i.e. rivers, lakes, ponds etc.)
Non-native (encompassing exotic, non-indigeneous & alien)	A species *introduced* outside its natural *range*.
Range	The geographical distribution of a species
Introduction	The deliberate or accidental release into the wild of a *non-native* species
Translocation	The human-assisted movement of fish within a specific water (i.e. rivers, lakes, ponds etc.)
Stocking	The release of a species into a specific water (i.e. river, lake, pond etc.) following its initial *introduction*
Establishment	The process whereby an *introduced* species reproduces and forms self-sustaining populations (i.e. not relying on further introductions)
Naturalisation	The process whereby an *established* species develops persistent populations
Dispersal	The natural dissemination (i.e. non human assisted) of a species from its point of *introduction*
Colonisation	The natural *dispersal* of an established population resulting in its *range* expansion
Spreading	A species expanding its *range* (can be human assisted or natural).
Invasion	The process whereby an *introduced* species has *established* populations, *spread* rapidly and presents a risk to *native* species
Lag phase	The delay between *introduction* and *invasion*

[4] Despite the desire to see the terms used for non-native introductions defined according to ecological processes only, when referring to non-native fish introduction this briefing paper encompasses both the ecosystem and national boundaries, as much of the published literature is set at this level.

2- ENTRY ROUTES AND MECHANISMS

The introduction of fish species into fresh waters is commonplace around the world. Common carp *Cyprinus carpio* L. appears to have been the first freshwater species transferred from its native range in eastern Europe, initially to Rome and then by monastic orders to other countries in Europe during the middle ages (Balon, 1974). Later, the development of aquaculture promoted the transfer of stock from the natural environment for growing on, and subsequently the transfer of fish from other continents. However, the large scale introduction of fish species into areas outside their native range is a comparatively recent activity (Fig. 1) (Welcomme, 1988).

Figure 1. Trends in number of species introductions into fresh waters by decade. (DIAS).

2.1 Drivers of freshwater fish introductions

Non-native species have primarily been introduced into new ecosystems through human activity, either deliberately or unintentionally (Vitousek *et al.*, 1997; Koo & Mattson, 2004). Information on the reasons for introductions are, however, far from complete, particularly pre-1900s, and the information available is often vague, indicating no obvious motive. The principal reported motives for introductions of non-native fishes into fresh waters (*n* = 2797) were aquaculture (39%) and improvement in wild stocks (17**%) (**Database on Introductions of Aquatic Species,

DIAS (access June 2008). Introductions for aquaculture have always been relatively important but came to the fore in the late 1900s with the development of salmonid aquaculture worldwide, and again in the 1960s and 1970s with the emphasis on tilapiine species and carps, especially in Asia (De Silva *et al.*, 2006).

Non-native species are reared locally in farms or imported from abroad for business reasons based on the most cost-efficient species in terms of production costs to output revenues, resistance to environmental stressors (e.g. pollution or parasites), and pre-existing knowledge of rearing methodologies and technologies. This includes the genetic improvement of farmed species, promotion of specific pathogen-resistant stocks and stock enhancement practices and ease of reproduction. Within this category, freshwater fishes such tilapias (*Oreochromis* spp.), Chinese and Indian carps and African catfish make huge contributions to aquaculture outside of their natural distribution range (De Silva *et al.*, 2006). All have led to a high number of escapes from aquaculture installations and the establishment of natural breeding populations.

2.1.1 Fisheries activities

The introduction of non-native fish species to modify wild stocks is an important fisheries management intervention that essentially responds to three perceived needs: 1) establishment of new fisheries; 2) enhancement of existing fisheries; 3) fill a 'vacant niche'. The main motives behind stock enhancement are the desire to improve fishery performance because the social and economic values of fisheries are huge, especially as demand for fishery products and improved leisure access increases. Indeed, Cambray (2003) suggests species introductions arising from recreational fishing practices are one of the principal causes of environmental degradation resulting in loss of biodiversity and that it requires a global solution. For example, total expenditure on recreational fishing across Europe currently exceeds € 25 billion per year (Arlinghaus *et al.*, 2002; Cooke & Cowx, 2006). Introductions of fishes for sport fishing include species (principally salmonids) valued for their sporting qualities and meat. Among these, the most widely dispersed are the rainbow trout *Oncorhynchus mykiss* (Walbaum), brown trout *Salmo trutta* L., brook trout *Salvelinus fontinalis* (Mitchill), and, among centrarchids, largemouth bass *Micropterus salmoides* (Lacépède). Similarly, North American catfish species, such as black bullhead *Ameiurus melas* (Rafinesque), brown bullhead *Ameiurus nebulosus*

(Lesueur), and channel catfish *Ictalurus punctatus* (Rafinesque), as well as the European catfish *Silurus glanis* L., have been introduced into ponds throughout Europe for sport fishing. Many fish species, including tilapias, carps and coregonids, have been introduced worldwide to create new fisheries, and enhance commercial yields or create subsistence fisheries, especially in impoverished water bodies. In some cases, this is to establish new communities, such as in newly-inundated reservoirs that have no indigenous pelagic species. Perhaps the most iconic examples are the introductions of Nile perch *Lates niloticus* (L.) into Lake Victoria to enhance fishery yield (Ogutu-Ohwayo & Hecky, 1991; Matsuishi *et al.*, 2006) and Lake Tanganyika sardine *Limnothrissa moidon* (Boulenger), into Lake Kariba or Lake Kivu (Marshall, 1995).

2.1.2 Bioengineering and biological actions

More recently, non-native fish species (6 %) have been intentionally introduced into open waters to manipulate the ecosystem through physical (bioengineering) or biological actions (predation of or feeding on another species) to control unwanted organisms (biocontrol; Ciruna *et al.*, 2004a), especially mosquito fishes *Gambusia affinis* (Baird & Girard) and *Gambusia holbrooki* Girard, for mosquito control (Kumar & Hwang, 2006), grass carp, *Ctenopharyngodon idella* (Valenciennes), for macrophyte control and silver carp *Hypophthalmichthys molitrix* (Valenciennes), for controlling phytoplankton. Whilst these introductions appear to have been highly successful, particularly relative to more costly and environmentally unacceptable alternatives such as insecticides and herbicides, the impact on the recipient ecosystems has yet to be fully evaluated, but evidence suggests the impacts can be detrimental. There is also a lack of certainty about the level of biological control that will be achieved and over what timescale (Wittenberg & Cock, 2001), causing considerable controversy over the concept of biocontrol as a management tool in freshwater ecosystems. For example, Chinese carps, native species of eastern Asia, have been introduced throughout the world to control excessive aquatic plant growth, but they have often failed to achieve their intended objective and displaced native fish species of greater commercial value (e.g. silver carp *Aristhchtys nobilis* - (Richardson) in the Danube Delta) or been the source of new parasite fauna that has affected native species (e.g. *Ergacilus* sp. from grass carp into UK waters, Cowx, 1997). In some cases, indiscriminate exploitation of aquatic vegetation has resulted in the destruction

of important habitats and food organisms (Cudmore & Mandrak, 2004). Similarly, mosquito fishes are aggressive foragers, feeding on a variety of prey, including the eggs, fry and larvae of native biota (Goodsell & Kats, 1999). Intentional introduction into the wild is fraught with risks if not regulated correctly because the invasive species and their associated fauna are introduced directly into open water bodies, and rarely held in aquaculture quarantine facilities.

2.1.3 The role of the ornamental fish trade

High volumes of ornamental fish are imported into western industrialised countries, especially from Southeast Asia, Africa and South America (Keller & Lodge, 2007). These non-native species are reared in local farms or imported from abroad for ornamental/aesthetics reasons, such as private or public aquaria and/or gardens, for example goldfish *Carassius auratus* (L.) and koi carp, cichlid fishes *Symphysodon discus* Heckel (discus), *Pterophyllum* sp. (scalar), and the poecilid fish *Xiphophorus* sp. (platy) (Copp *et al.*, 2005b). It is unlikely that many of these fishes could survive and spread as invasive alien species in temperate waters because of their particular ecological and physiological requirements (warm water >15 °C), although goldfish is an exception. However, some species find ideal conditions in lower latitudes (e.g. southern Europe or the southern USA) where water temperatures do not fall below the lower temperature thresholds, or are able to tolerate the ambient conditions. Furthermore, some species find refuge in waters artificially warmed by effluent discharge, for example downstream of power stations and other industrial facilities.

2.1.4 The role of research activities

Several species have also been imported and stocked live for use in research. This is mainly for new, non indigenous species imported for experimental aquaculture trials, e.g. crossbreeding with native and non-native fishes (Colombo *et al.*, 1998) or for biocontrol assays or other research issues, e.g. bio-essays conducted with alien species to stop bioinvasion by other alien species. Generally, the risk of dispersal is minimal because the research institute applies quarantine measures to experimental trials, which are conducted in closed systems and not in open waters (Colombo *et al.*, 1998).

2.1.5 Accidental introductions

Perhaps the category that causes greatest concern is accidental introductions. Nearly 8 % of introductions are accredited to this mode. Although not specified, escape from aquaculture installations is probably a main cause, and this in turn has led to diffusion along water courses. Similarly dispersal through ballast water disposal is another mechanism contributing to accidental introductions (see section 2.2) With the extensive development of inter-catchment transfers of water resources that now take place, it is likely that dispersion of species by this mechanism will increase in the future. Accidental introductions that result from contaminants within batches of other fishes that are being deliberately imported or released are particularly problematic. For example, recreational anglers have been responsible for the direct translocation of fish species (West *et al.* 2007), including the transfer of carp either by accident, ignorance or deliberately to new fishing areas (McDowall, 1996; Koehn, 2004). The Ponto-Caspian cyprinid, topmouth gudgeon *Pseudorasbora parva* (Temminck & Schlegel), which was initially discovered in Europe in a fish farm facility in Romania in 1960, is presumed to have been introduced as a contaminant of a consignment of Chinese carps and is a prominent example of an accidental introduction as it is now found throughout Eastern, central and western Europe (Holcík, 1991; Gozlan *et al.*, 2002). The use of live fin-fish as live bait can also lead to the introduction of non-native alien species (Lintermans, 2004; Kerr *et al*., 2005) and examples include introduced ruffe *Gymnocephalus cernuus* (L.) into parts of northern England outside its native range (Drake, 2005). Similarly, the enrichment of the Irish cyprinid fauna has been related to the release of bait species by recreational anglers targeting pike *Esox lucius* L. (Caffrey *et al*., 2008). These mechanisms likely contribute to indiscriminate, inter-river basin dispersal and potentially account for the rapid expansion of the range of many non-native species.

2.1.6 The role of aquaculture

In all the above cases, aquaculture plays a major role in the provision of stocking material. This is either through artificial rearing of the material or acting as growing-on facilities prior to fish being stocked into the wild. The propensity for impact from non-native species is high in these activities, not least because the species are stocked directly into open or semi-enclosed environments. Consequently, the risks of dispersal of the target species are high, as well as those of accompanying non-target species.

There is potential for the transfer of new pathogens, viruses, bacteria, fungi, parasites and other organisms to new areas where they may not be pathogenic under normal environmental conditions for native species. The risks are also high because aquaculture facilities play an intermediary role when novel species are brought into a country before being stocked out. If health checks and inspections are weak, and quarantining not used, then the risks of transfer of pathogens or contamination of material by alien invasive species are high.

2.1.7 Summary

Drivers of use of non-native freshwater fish species are essentially direct interventions by humans, but they should also include upstream and downstream activities allied to fisheries and the potential for invasion and subsequent colonisation of already established populations. In the former case, it should be recognised that these drivers operate both within and outside the law. There are many instances where fishes are moved illegally, but always for one of the purposes outlined above (Minchin & Rosenthal, 2002).

2.2 Colonisation and invasion routes

In terms of colonisation and invasion routes, a number of natural and human-mediated mechanisms and routes are known. For example, in Europe, there are effectively three invasion routes that are responsible for the dispersal of aquatic organisms, although not principally for freshwater fishes (Panov *et al.*, 2007). These are: the Northern Corridor covering the route from the Volga River to the Baltic Sea via Lake Beloye, Lake Onega, Lake Ladoga and the River Neva; the Central Corridor covering the route from Dnieper River to the River Rhine via Vistula River, Oder River & Elbe River; and the Southern Corridor covering the route from the Danube River to the Rhine River (de Vaate *et al.*, 2002; Karatayev *et al.*, 2008). Each consists of an interlinked network of 30 main canals, with more than 100 branches and more than 350 ports.

Shipping is a further potential mechanism contributing towards the dispersal of non-native species, and accounts for 25 % of the recorded introductions of freshwater organisms in Europe (Gollasch, 2007), mainly via ship ballast water and hull fouling

(Bax *et al.*, 2001). It is estimated that 10 000 or more species of marine organisms may be transported around the world in ships' ballast water each week (Carlton, 1999), but movements of freshwater fishes are less common. Prominent examples are the introductions of ruffe and round goby *Neogobius melanostomus* (Pallas) into the Great Lakes of North America (Scott & Crossman, 1973; Grigorovich *et al.* 2003; Holeck *et al.* 2004).

3. ECOLOGICAL IMPACT OF NON-NATIVE FRESHWATER FISH INTRODUCTIONS

In the last decade, the profile of non-native species has been strongly advocated as a driver of biodiversity loss but 'there is a growing minority of ecologists who question whether there is strong evidence that non-native species are a direct cause of native population decline' (Didham *et al.*, 2005). Often, non-native species introductions are correlated with other drivers of environmental change, such as habitat modifications (Moyle, 1986; Brown & Moyle, 1991, 1997; Kitchell *et al.*, 1997; Tejerina-Garro *et al.*, 2005; McDowall, 2006) and management practises (Bain, 1993; Wootton, 1994; Whittier & Kincaid, 1999; Caissie, 2006; Lewin *et al.*, 2006). It remains very difficult to distinguish the primary cause of environmental change and what is actually co-occurrence, which by definition represents an association (Didham *et al.*, 2005).

A further issue is the lack of common understanding of what actually represents an ecological impact (Simberloff, 2003; Sagoff, 2005; Brown, 2007; Sagoff, 2007; Simberloff, 2007; Gozlan, 2008). For some, the mere act of introducing a non-native species into an ecosystem is a source of ecological impact (i.e. guilty before proven innocent; Simberloff, 2007), whereas for others the need to define this nebulous and ill-defined concept of ecological impact is fundamental to progress contemporary thinking and develop more robust policies on non-native species (Sagoff, 2007).

3.1 What does ecological impact mean in the context of fish introduction?

Freshwater ecosystems and their fish assemblages present a particularity since drainage basins may be considered "biogeographic" islands. The obstacles to natural fish migration between hydrographic catchments over large temporal scales imply that extinction and speciation processes are largely specific to each basin. In this context, understanding of what constitutes a harmful biological introduction is crucial to the preservation of freshwater ecosystems and their biodiversity. There are several key aspects often considered in association with non-native fish introductions: predation, habitat degradation, competition for resources, hybridisation and disease transmission. However, according to a recent report to the European Commission[5] and in agreement

[5] Committee for Fisheries and Aquaculture 4-3.03.2008, ref. D 01960. Point 4 of the agenda: "(Information and discussion) Draft Commission Regulation laying detailed rules for the

with Gozlan (2008), consideration of these aspects in isolation is insufficient to characterise the ecological impact of an introduced fish. Instead, there is a requirement for the above key aspects to be associated with a quantifiable and significant decline of biological or genetic diversity threatening long term integrity of native species. The question is not about ecological changes as changes are inevitable when any species is introduced and established in a naïve ecosystem, but rather if these changes lead to a **measurable loss of diversity or change in ecosystem functioning**. Only in this case can fish species introduction be considered harmful.

3.1.1 Impact through predation

Foraging habits are often one of the first ecological aspects studied for any introduced non-native fish (Brown & Moyle, 1991; McIntosh & Townsend, 1995; Kitchell *et al.*, 1997; Shurin, 2001; McDowall, 2006; Weyl & Lewis, 2006; Bampfylde & Lewis, 2007; Yonekura *et al.*, 2007). Foraging, however, is a difficult ecological aspect to characterise in the context of overall ecological impact. Although diet overlap and prey-predator relationships are strong indicators of species interactions and of the trophic position attributed to the introduced species, these are not necessarily indicators of loss of biological diversity or profound changes in ecosystem function (Isumbisho et al. 2006). The real tipping point indicator of a significant ecological impact in the system is when gradual changes in environmental parameters resulting from non-native interactions lead to discontinuous changes in ecosystem functioning. This is characterised by the loss of ecosystem resilience to disturbances and the loss of its capacity to maintain its functioning, structure, various feedbacks and service provisions (Folke *et al.*, 2004). Ecosystems with less functional duplication are more vulnerable to disturbance (Schindler, 1990), and in some cases introduced fish could diminish the overall resilience of the ecosystem through modification of the trophic structure (Folke *et al.*, 2004). However, most studies on non-native introductions only identify initial effects of alteration (Moretto *et al.*, 2008; Smith *et al.,* 2009), cover a relatively short period (Gozlan *et al.*, 2003; Encina *et al.*, 2004; Jordan *et al.*, 2004; Buria *et al.*, 2007), or do not explicitly identify effects on other ecosystem processes (Zaret & Paine, 1973; Bradford, 1989; Nelva, 1997) or changes in ecosystem function

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implementation of Council Regulation (EC) No 708/2007 concerning use of alien and locally absent species in aquaculture".

(Xie & Chen, 1999; Moyle & Davis, 2000; Bernardo *et al.*, 2003; Holcik, 2003; Russell *et al.*, 2003; Olsen & Belk, 2005). Even studies that characterise interactions between non-native and native fauna rarely discuss these processes in the global context of ecosystem functioning (McIntosh & Townsend, 1995; Chapleau *et al.*, 1997; Funk & Dunlap, 1999; Ling, 2004; Maezono *et al.*, 2005).

A quantitative understanding of long-term changes occurring in food-web structure and functioning is a good way to characterise the long-term implications of non-native fish introductions (Ogutu-Ohwayo & Hecky, 1991; Vander Zanden *et al.*, 2003; Eby *et al.*, 2006). The increasing use of stable isotopes (e.g. ^{15}N, ^{13}C) to characterise food web structure now allows their routine use for long-term monitoring of changes in ecosystem functioning that result from introductions (Vander Zanden *et al.*, 1999). Several studies have integrated this methodology into the study of non-native ecological impacts and provide useful insights into the role played by various fish introductions to the overall ecosystem (Buktenica *et al.*, 2007). Introductions of fish can alter food web structure and ecosystem functioning through top-down or bottom-up mechanisms, and/or through consumer and resource co-limitation (Eby *et al.*, 2006). Outcomes range from changes in trophic efficiency (Matyas *et al.*, 1998; Parker *et al.*, 2001; Shurin, 2001; Angeler *et al.*, 2002; Isumbisho *et al.*, 2006; Reissig *et al.*, 2006) to changes in biogeochemical cycling in the aquatic system itself (Kyle *et al.*, 2001; Lung'ayia *et al.*, 2001; Schindler *et al.*, 2001; McIntyre *et al.*, 2007). But this could also include the flux of nutriments with the riparian terrestrial ecosystem (Cole *et al.*, 2000; Duarte & Prairie, 2005), which could lead to overall changes in the resilience of ecosystems.

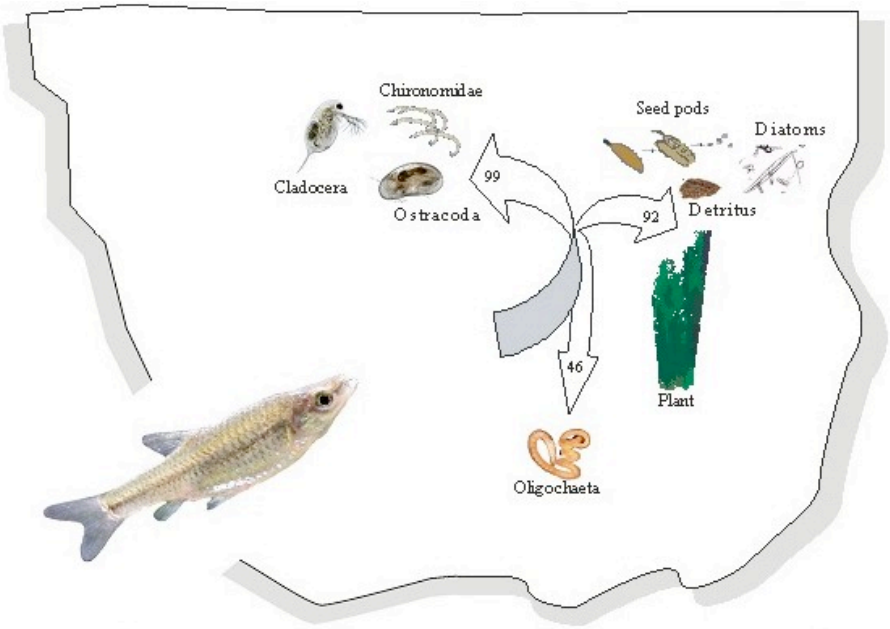
3.1.2 Impact through competition

Whilst shifts in the trophic structuring of food webs may be relatively easy to characterise, more subtle changes resulting from competition between species of similar trophic level can also occur following an introduction. For example, brown trout introduced to New Zealand streams have out-competed and displaced native top predator species of the Galaxiidae family, with their distribution now restricted to trout-free reaches (McDowall, 2006). In addition, through changes of invertebrate and algal production, significant changes in nutrient cycling and, in particular, nitrate were identified (Simon *et al.*, 2004), although the full implications of these changes in the

functioning of the ecosystem remain unclear. However, trophic interactions of this type are not always apparent following the introduction of a non-native fish, as shown in a study of competition between rainbow trout and Atlantic salmon *Salmo salar* L., where intra-specific competition was found to be greater than inter-specific (Hearn & Kynard, 1986 in Fausch, 1998).

It is also important to consider that quantitative measures of *in situ* competition are complex and often influenced by other factors, including temperature and habitat (Gurevitch *et al.*, 1992; Fausch, 1998; Potapov & Lewis, 2004). Nonetheless, if non-native fish introductions were to have a significant ecological impact on native species and ecosystems through interspecific competition, then an unequivocal measure of this impact should be measurable and would need characterisation (see Box 1). Despite the importance of laboratory experimentation in highlighting competitive interactions (Caiola & Sostoa, 2005; Zimmerman & Vondracek, 2006), field studies are key in providing a more holistic vision of non-native species induced ecological impact emerging from competitive forces (Blanchet *et al.*, 2007).

Box 1. Effect of topmouth gudgeon *Pseudorasbora parva* introduction on native fishes.
Topmouth gudgeon, which originates from Asia is the most successful invasive fish in Europe since its first introduction in 1961 in the lower part of the Danube. It is a small cyprinid species with early maturity and batch spawning females. The male defend a primitive nest where eggs are guarded until hatching. Where introduced, this species has generally established extremely large populations, which compete with native species for food resources. As seen in the example below in water bodies where topmouth gudgeon was introduced a shift in the diet of native species has been observed using stable isotopes analysis (δ^{13} C and δ^{15} N). However, despite observing a significant shift in the native species diet, it is too early to appreciate fully the ecological consequences of such a shift on the ecosystem functioning.



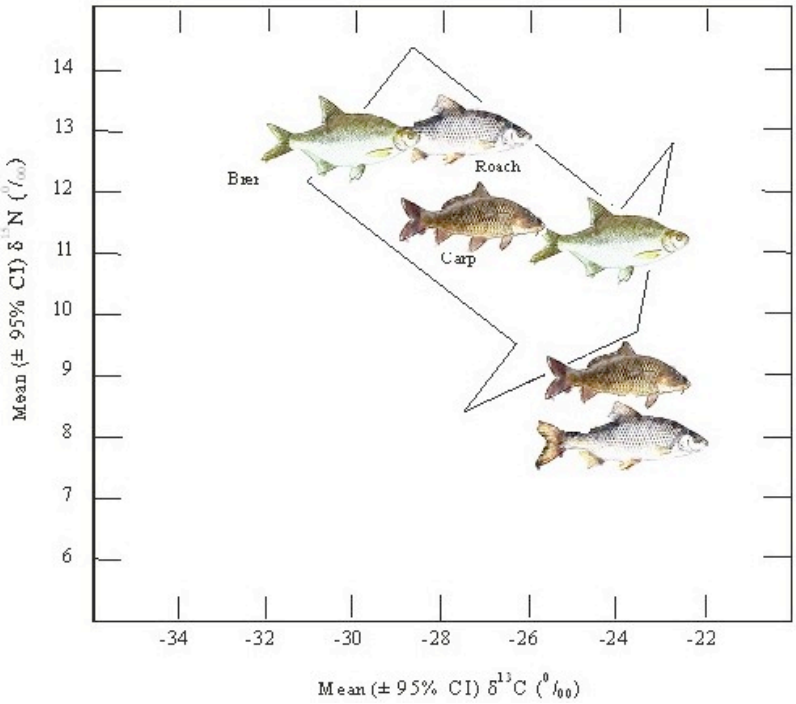


Figure 1. Top: Typical diet spectrum for topmouth gudgeon. Numbers indicates the frequency of occurrence (%). Bottom: Shifts in δ^{13} C and δ^{15} N values of native fish species diet after the introduction of topmouth gudgeon.

3.1.3 Impact through hybridisation

Despite hybridisation being a common phenomenon in fishes (i.e. more than in other vertebrates), loss of genetic integrity is often reported as a potential ecological impact associated with the introduction of non-native fishes. Hybrid zones, characterised by relatively small areas where genetically distinct species meet and mate, are often exacerbated by limited spawning habitat, are common in freshwater ecosystems and are synonymous with clines (Scribner *et al.*, 2001). Hybridisation does not necessarily result in the loss of species diversity, as illustrated by the colonisation by the non-native nase *Chondrostoma nasus* (L.) through a man-made canal into the Rhone catchment where the French nase *Chondrostoma toxostoma* (Vallot) occurred naturally. These two species have hybridised for decades creating a "hybrid zone", whilst at the same time maintaining pure bred populations within the same hydrosystem (Costedoat *et al.*, 2004; Costedoat *et al.*, 2005).

Although the process of hybridisation has been greatly enhanced by human activities especially through aquaculture, habitat modifications and fish introductions (Scribner *et al.*, 2001), the link with the loss of global fish biodiversity is still rarely demonstrated. Hybridisation that results from non-native fish introductions may be considered as an additional stressor on already fragile populations of species that have a limited distribution and could, in some cases, lead to loss of genetic integrity (Allendorf *et al.*, 2004; Hanfling *et al.*, 2005; D'Amato *et al.*, 2007). The great difficulty is to put these local hybridisation events in the context of global fish species conservation. For example, hybridisation between crucian carp *Carassius carassius* (L.) and the introduced goldfish and common carp is commonplace (Hänfling *et al.*, 2005), affecting 38 % of crucian carp populations in the UK. However, it is still unclear what this figure represents in the overall global distribution of crucian carp and if it represents a significant threat to the conservation of this species. It is also difficult to discriminate between the importance of the time of introduction and habitat availability in these hybridisation processes, as it may explain, for example, why levels of hybridisation between crucian cap and common carp vary significantly between locations (Pullan & Smith, 1987; Hanfling *et al.*, 2005). Hybridisation related to non-native fish introductions only accounts for 17 % of known hybridisation (Scribner *et al.*, 2001) and around 4 % of all non-native fish introductions in the world. Therefore, this issue is relatively limited when looking at ecological impacts

associated with global non-native fish introductions, although it could be greater at the local scale (Allendorf, 1991; Allendorf *et al.*, 2004; D'Amato *et al.*, 2007). In some cases, local adaptations could be lost through introgressive hybridisation with introduced species and could potentially decrease their chance of long-term persistence (Allendorf *et al.*, 2004).

3.1.4 Impact through habitat modification

Among non-native fish introductions, those that may have the most severe ecological consequences are engineer species that directly modify their ecosystem, generating cascading effects (Crooks, 2002). These impacts are often achieved through a change in physical habitat and consequences are typically proportional to long term stability and complexity of the ecosystem (Brown & Moyle, 1991; Power, 1992; Fuselier, 2001; McDowall, 2006). A change in habitat, and particularly macrophytes or phytoplankton community/biomass, would typically result in a substantial modification of ecosystem function, with long term implications for many species and the overall integrity of the ecosystem.

This is well illustrated by the world-wide introduction of common carp (Khan *et al.*, 2003; Koehn, 2004; Pinto *et al.*, 2005), a species that modifies aquatic ecosystems through its foraging behaviour, resulting in uprooted plants and the re-suspension of sediments. The knock-on effect is increased turbidity, which prevents plant re-growth and impacts on phytoplankton biomass. Such trophic cascade is largely influenced by the density of carp and the type of macrophytes present (Miller & Crowl, 2006), and produces direct effects on macrophyte, invertebrate and plankton diversity, and indirect effects via changes in water chemistry, turbidity and wind effect. Despite the recorded ecological impacts of carp on habitat, the species is now widely distributed, with populations in Northern Europe principally maintained by regular stockings, as their reproduction is inhibited because of the requirement for sustained temperatures above 20 ^{o}C (McDowall, 1996). Habitat modification of this type is also observed, albeit to a lesser degree, following the introduction of other fish species, both within and beyond their native ranges (e.g. common bream *Abramis brama* (L.)), but is mainly associated with benthivores.

In the same way that a non-native 'engineer' fish can modify habitat, habitat itself could be engineered to favour native species to the detriment of those introduced, or to at least promote their co-existence. For example, following the introduction of western mosquito fish in Utah, USA, populations of the native least chub *Iotichthys phlegethontis* (Cope) declined, mainly through habitat degradation but also as a result of the introduced mosquito fish that utilise the same spawning habitat (Ayala *et al.*, 2007). A detailed field study of habitat use of these two species, however, highlighted subtle differences resulting from the evolutionary history of both species and characterised the need for habitat management to promote deeper, cooler habitats that would support the coexistence of the species, as opposed to warm, shallow, marsh habitats that favour western mosquito fish (Ayala *et al.*, 2007).

Pro-active habitat management is a powerful tool with significant implications for ecosystem function and should not be under-estimated as an effective tool to limit the impact of introduced non-native fishes (Maret *et al.* 2006; Brown & Moyle, 1991). In some cases, such as the least chub, habitat management could be used to enhance pockets of conservation interest targeted to a particularly vulnerable native species or group of species (Ayala *et al.*, 2007). In many places, habitat changes are happening whether non-native fish are introduced or not, and the desire to maintain or restore a pristine habitat in the sense of original habitat all too often is reminiscent of Plato's myth of the cave where the vision of this pristine lost environment provides a distorted view of a changing reality.

3.1.5 Impact through disease transmission

Aquaculture has been the main source of non-native fish introductions worldwide (Bartley & Subasinghe, 1996; Blanc, 1997; 2001; Gozlan, 2008) and has been followed by increased introduction and translocation of infectious pathogens in freshwater ecosystems (Blanc, 1997; Gozlan *et al.*, 2006). The strong correlation between source of host and source of pathogen indicates that aquaculture not only facilitates disease emergence but may also act as a source of pathogen introduction. However, this account is possibly biased towards pathogens of aquaculture species due to limited knowledge of the pathogenic fauna of wild fishes. Many fish disease specialists believe that introduced hosts act as a reservoir population from which infection can 'spill-over' to sympatric wildlife. Pathogen–pollution, which would

otherwise fail to persist, instead underpins the emergence of disease in naïve populations (Daszak, 2000; Gaughan, 2002; Gozlan *et al.*, 2005). Although there are numerous examples of disease emergence after species introduction, there are likely to be many more that have not been identified. Epidemics in the wild that cause high mortality or changes in fish behaviour are easily characterised, but are less easily identified when they cause long-term gradual population declines. It is relatively complex to link fish population decline with pathogen infection in wild populations because diseased or dead animals are rapidly consumed by predators or necrophages. This is one of the reasons why the impact of an introduced pathogen might very often go unnoticed. This link should become more obvious when developments in sampling protocols and diagnostic techniques enable better characterisation of emerging infectious pathogens (Bull, 1994; Vanderploeg *et al.*, 2002; Gozlan *et al.*, 2005; Murray & Peeler, 2005).

The discovery of an infectious agent implicated in the decline of the endangered sunbleak *Leucaspius delineatus* (Heckel) has highlighted the potential for the spread of fish pathogens via the introduction of non-native invasive fish species. In this case, the topmouth gudgeon which was found to be a healthy host of the rosette agent *Sphaerothecum destruens* (Gozlan *et al*. 2005), a microparasite identified in North America as affecting a large range of salmonid species, particularly Chinook *Oncorhynchus tshawytscha* (Walbaum) and Atlantic salmon. The pathogen causes a chronic disease in sunbleak and has been related to the decline of sunbleak across Europe and the concomitant spread of topmouth gudgeon (Gozlan *et al.*, 2005). This is typically the worst case scenario where a non-species specific pathogen is associated with an invasive healthy carrier and is allowed to be associated with the aquaculture trade, thus accelerating its further dispersal. Despite causing mass mortality, this pathogen is still difficult to characterise in wild populations as a result of the chronic nature of the infections but nonetheless appears to pose a substantial threat to fish biodiversity (Gozlan *et al.*, 2005).

The introduction of *Anguillicolloides crassus* in Europe in the early 1980s, is another compelling example of a disease being spread as a result of global non-native fish movements and introductions for aquaculture purposes (Kirk, 2003; Ashworth & Blanc, 1997). Originating from Asia, *A. crassus* is a nematode that can severely

impair swimbladder function and has been causing serious losses in intensive European eel *Anguilla anguilla* (L.) production. An expanding eel trade has promoted the global distribution of the parasite in Europe and North Africa as well as in the eastern part of North America, where infected American eels *Anguilla rostrata* (Lesueur) have been found (Kirk, 2003). The majority of *A. crassus* records come from aquaculture facilities as it is easier to detect dead or diseased eels in cultured populations. However, it seems that in the few cases of host-mortality reported in wild populations (Ashworth & Blanc, 1997), high losses resulted from a combination of high-density eel stocks and adverse environmental stressors (i.e. high temperatures and low dissolved oxygen levels).

Owing to diagnostic limitations, it remains difficult to clarify the extent of pathogen introductions associated with non-native fish introduction. However, Torchin *et al.* (2003) specifically tested the hypothesis that hosts were less parasitised in their introduced range than those in their natural range, leading to demographic release in the host population and facilitating invasion. It was concluded that this may be as a result of reduced probability of introduction (few host population founders), absence of intermediate host in the introduced range and host specificity limitations; non-native fish were less burdened with parasites in the novel environment (Torchin *et al.*, 2003).

Few studies have been conducted on the impact of pathogens on free-living hosts. Further, the number of studies examining the role of disease on structuring fish populations is even more limited, particularly in relation to introduced pathogens. Those pathogens that are studied tend to be ones that impact immediately and negatively on host population dynamics, usually in the form of epidemics. Future pathogen-related issues are likely to come from aquaculture farming in semi-open systems, with these facilities increasing the probability of introduction through high densities of hosts and their frequent introduction. These types of facilities are the main entrance of non-native fish introductions and intensive aquaculture conditions such as high stocking densities, accumulation of waste, handling and poor water quality all serve to compromise immunity and favour disease emergence (Blanc, 2001; Gozlan *et al.*, 2006).

3.2 What is the global ecological cost of non-native fish introduction?

A principal concern regarding non-native fish introductions is uncertainty about whether a particular introduction into a particular ecosystem will have no detrimental impact on its ecology and the long term sustainability of its biodiversity (Clavero & García-Berthou, 2005; Simberloff & Stiling 1996b). Even when a species has been introduced for decades with no obvious impact, the existence of a lag phase following introduction means that an impact may still become apparent subsequently. Many conservation ecologists believe that ecological impacts following non-native fish species are unavoidable and even if it seems that there are no impacts, this is because they have yet to be quantified (Ricciardi, 2004; Simberloff, 2003). It may be considered easier to demonstrate a correlation between ecological changes and non-native fish introductions rather than the absence of changes, and according to the 'guilty before proven innocent approach', there is always the possibility that changes have occurred but are yet to be detected (Simberloff, 2007). This is the most commonly accepted view among scientists with interests in biological invasions, but has a tendency to skew observations towards introduced fishes generating ecological problems and serves to reinforce the notion that outcomes of non-native introductions are generally detrimental (Brown, 2007; Gozlan, 2008; Sagoff 2007). It also means that in the management of introduced fishes, it is difficult to prioritise efforts to those species that are most ecologically damaging, as the majority have to be considered as detrimental to receiving ecosystems (Gozlan, 2009).

An inherent issue in this argument is deriving a common understanding of the term 'ecological impact' (Gozlan 2008; Parker *et al*., 1999). It is intuitive to consider that the integration of a species in an ecosystem will induce changes in the system and its biodiversity, particularly as the introduced species commences its basic biological functioning (foraging, reproduction and habitat utilisation). If the measure of ecological impact is based on the characterisation of changes in the system, then it is correct to assume that ecological impacts following introductions of non-native fishes are unavoidable. The question is whether this is a realistic approach to the problem or whether should there be more discrimination when looking at the impacts of non-native fish introductions. Given that many freshwater ecosystems are undergoing radical changes because of anthropogenic pressures such as habitat degradation, impoundments, gravel extraction, change in land use and climate change (Francis &

Schindler, 2006; Hoagstrom *et al*., 2008; Kruk, 2007; Seilheimer *et al*., 2009), then it is against this back-drop that the impact of introduced non-native fishes should be measured.

In a recent review of the demonstrated impacts of non-native fish introduction worldwide, Gozlan (2008) came to similar conclusions as Simberloff (2007) and the 'tens rule' of Williamson (1996). This predicted that 10% of all introductions will become established and that 10% of those established will become pests. Thus, for every 100 non-native fish species introduced, 99 will not be associated with serious ecological impacts. Notwithstanding, when an ecological problem occurs at a meso-scale, there could be serious consequences for local biodiversity. This is seen with the introduction of the Nile perch in Lake Victoria (Goldschmidt *et al.*, 1993; Kitchell *et al.*, 1997; Pringle, 2005) and brown trout in New Zealand (Townsend, 1996; McDowall, 2003). However, biological and behavioural differences between family and species of fish allow some level of discrimination between those non-native fish introductions presenting a high risk of ecological impact and those that are likely to prove relatively benign (Kolar & Lodge, 2002; Gozlan, 2008). It is not as though the 1 % of species presenting ecological risk is random and so tools are being developed to screen impacts of future non-native introductions (Section 5). In the meantime, future efforts to limit losses of global aquatic biodiversity resulting from non-native fish introductions need to be focused on aquatic areas of high conservation value that are also characterised by a high risk of non-native species introduction. These could easily be identified because the major pathways for fish introductions are known (Section 2) and are related to some human activities (Copp *et al.*, 2007).

4. ECONOMIC IMPACTS ASSOCIATED WITH THE INTRODUCTION OF NON-NATIVE FRESHWATER FISHES

4.1 Economic and social benefits

An aspect associated with the movement of non-native species that is often neglected are the socio-economic implications of their use and spread. Economic activity, particularly globalization through trade, is the fundamental human cause of non-native introductions (Perrings *et al.*, 2000; Perrings *et al.*, 2002; Pimentel, 2002; Koo & Mattson, 2004; Taylor & Irwin, 2004; Pimentel *et al.*, 2005). The general consensus is that non-native invasive species have a negative economic impact. This is, however, very much a misconception as there are both direct (e.g. revenue and food security v cost of eradication and control and loss of biodiversity) and indirect (wider benefits to society v loss of ecosystem goods and services), positive and negative, social and economic impacts of introduced non-native species. The typical picture painted can be summarised by the higher number of ecologically adverse impacts of non-native species than beneficial (Table 2). If the socio-economic dimension is reviewed, this picture is potentially very different. Unfortunately, the economic implications of species introductions have rarely been evaluated because of problems associated with determining environmental costs arising from a lack of adequate data and approaches for meaningful comparisons; for example, how to value ecosystem services and products. Information is often fragmented or anecdotal, but the literature is replete with evidence of the positive and negative impacts of non-natives. For example:

- approximately 17 % of the world's finfish production is due to non-native species;
- production of the African cichlid tilapia is much higher in Asia (>1.6 million t in 2005) than in most areas of Africa (245 000 t);
- introduced salmonid fishes in Chile support a thriving aquaculture industry that is responsible for approximately 30 percent of the world's farmed salmon and directly employs approximately 30 000 people.

4.2 Economic and social costs

The economic and social benefits accrued from non-native fish introductions must be weighed up against the environmental as well as the detrimental social and economic impacts of introduction of non-native fishes. Aquatic ecosystems may be

affected by the introduced species through predation, competition, mixing of exotic genes, habitat modification and the introduction of pathogens (Section 3). Human communities may also be impacted through change in fishing patterns because of a newly-established fishery or through changes in land use and resource access when high valued species are introduced into an area. Pimental *et al*. (2000, 2005), and the references and estimates therein, provided examples of these issues, although mainly for the USA. For example, 138 non-native fish species have been introduced into the United States (Courtenay *et al.*, 1991; Courtenay, 1993; 1997; Dill & Cordone, 1997), most of which have established in states with mild climates, such as Florida (50 species; Courtenay 1997) and California (56 species; Dill & Cordone 1997). In Hawaii, 33 alien freshwater fish species have become established (Maciolek, 1984). As a consequence, 44 native species of fish are now threatened or endangered by non-native, invasive fishes and an additional 27 native fish species are also negatively affected by introductions (Wilcove & Bean, 1994). One assumes that the other human impacts on these species have been accounted for prior to attributing all the blame to non-native species.

Despite positive gains, as illustrated with sport fishing supported by introduced fishes contributing $69 billion annually to the economy of the United States (Bjergo *et al.*, 1995), the conservative economic losses due to non-native fishes are estimated to be around $5.4 billion, annually. These figures are comparable with wider assessments of environmental and economic damages caused by a wide range of alien invasive species, which have been estimated at between a total of $97 billion for 79 harmful non-native species over the period between 1906 and 1991 (OTA, 1993) and $137 billion per year from a USA study using a larger number of invasive species, including terrestrial taxa (Pimentel *et al.*, 2000; Pimentel *et al.*, 2005). Despite these figures, there is a lack of precise data on the economic costs of non-native invasive species in inland fisheries. Boxes 2 and 3 provide two more in depth examples of the outcomes of the introduction of fish species, *viz.* the introductions of Nile perch into Lake Victoria and rainbow trout worldwide. These case studies provide a review of the economic and social benefits but, as with other studies, fail to provide costs of the loss of ecosystem goods and services.

One aspect that is not covered by these examples is the high cost of eradication or control of non-native species should they exhibit negative impacts on the recipient ecosystem (Shaw & Seiger, 2002; Section 6). Once established, invasive species can be extremely difficult and costly to control or eradicate. For example, after opening up of the St. Lawrence Seaway, sea lamprey *Petromyzon marinus* L. has dispersed through the Great Lakes of North America, impacting on fisheries that generate up to $4 billion for the region economy annually, offering recreational angling opportunities for five million people and providing 75 000 jobs (Lovell & Stone, 2005). To protect these fisheries, a number of control methods for lampreys have been in force for many years, including lampricide for larvae control, barriers, traps and a sterile male release programme (Great Lakes Fishery Commission, 2004). Estimates for this control action vary but are likely in the region of $10 million spent annually for control and research and another $10 million on fish stocking (Lovell & Stone, 2005). Without this control, lost fishing opportunities and indirect economic impacts are estimated at $500 million annually (OTA, 1993).

It is often argued that prevention, control and eradication of invasive species should be international, and most certainly a global public good because of the high costs involved (Perrings *et al.*, 2002). There is, therefore, a need for optimal policies regarding invasive non-native species that minimise the likelihood of their entry (or for deliberately introduced species, their escape into unintended habitats), taking into account the costs of prevention, and policies that minimise the damage they consequently cause, bearing in mind costs of control or eradication. However, there are challenges with efforts to control and/or prevent invasive aquatic species and these include; the high level of scientific uncertainty surrounding: i) the likelihood of species entry; ii) their invasiveness; and iii) the identification of specific economic, environmental and/or social impacts potentially caused by the species.

Table 2. Effects of introduced fish on ecological and socio-economic (in brackets) environments by reason for the introduction. Data represent number of records from FishBase. Other includes accidents, bait, forage, to fill niche, research and diffusion.

Impact	Fishing	Aquaculture	Ornamental	Bio-control	Unknown	Other
Adverse	36 (2)	78 (8)	17 (5)	23 (9)	13 (0)	40 (12)
Beneficial	16 (87)	52 (283)	11 (42)	11 (19)	3 (10)	6 (15)
Unknown	28 (16)	76 (49)	9 (9)	8 (2)	21 (3)	
Not indicated	196 (299)	949 (815)	169 (150)	106 (122)	459	283

Box 2. Positive and negative impacts of the introduction of Nile perch *Lates niloticus* (Linneaus) in Lake Victoria

The Nile perch was introduced into Lake Victoria from Lakes Albert and Turkana during the 1950s and 1960s to compensate for depleting commercial fisheries by converting low value haplochromines into higher value and more easily captured fish (Fryer, 1960; Ogutuohwayo, 1990; Ogutuohwayo & Hecky, 1991). After a 20-year lag period, their abundance exploded in the fishery. At its peak, *L. niloticus* was contributing more than 200,000 t of fish (Fig. 1), worth in excess of US$ 250 million, to the national economies of the riparian countries, Kenya, Tanzania and Uganda, through export of fish products, mainly chilled and frozen fillets. This brought added benefits in terms of employment for over 1 million people in the capture, processing and ancillary support services sectors.

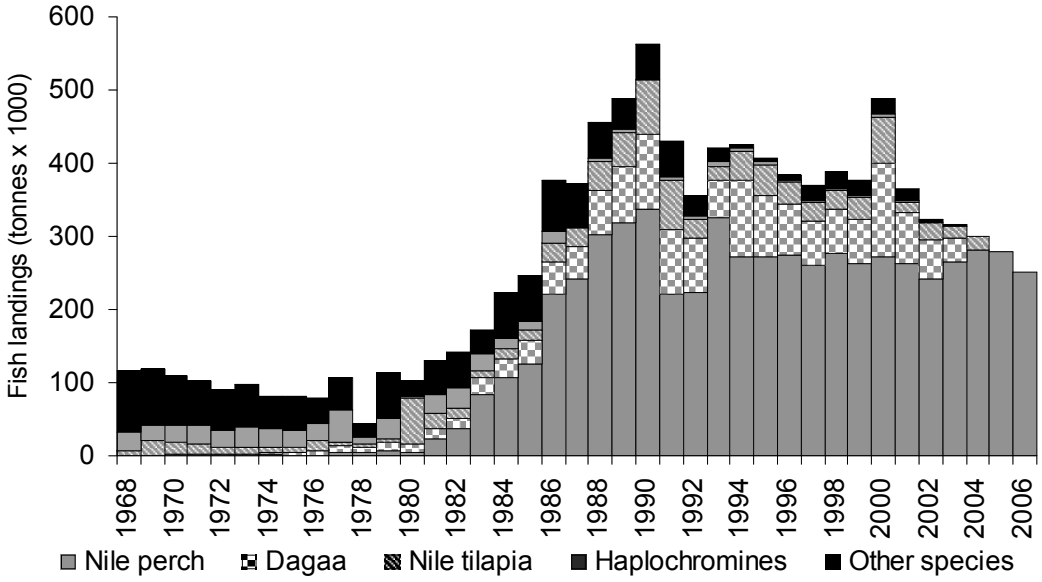


Figure 2. Landings (tonnes) of the main commercial fish species from Lake Victoria between 1968 and 2006. Note data for species other than Nile perch were not available from 2004.

There were also considerable secondary benefits, including infrastructural development in terms of new roads, electricity and communications, as well as schools and medical facilities, all benefiting the rural poor fishing communities. These gains must, however, be balanced against disruption of lake ecosystem (Kolding *et al.*, 2006) and loss of biodiversity (Balirwa *et al.*, 2003). Many blame the loss of several hundred species from the endemic halpochromine species flock on the introduction of the Nile perch (Witte *et al*., 1992). In reality, these species were being overfished from as early as the 1920s and this contributed as much to the demise of the species (Kolding *et al.*, 2006). The Nile perch possibly took advantage of reduced predation by the larger, piscivorous haplochromines that were first fished out and exploded in the late 1970s and 1980s (Fig. 1) (Matsuishi *et al.*, 2006), enabling their rapid increase in abundance. This enabled development of a major *L. niloticus* fishery in which fish were initially processed by smoking. As a consequence, much of the riparian vegetation in the region was stripped for the fires and this, coupled with a huge population explosion partly associated with the race for fish, resulted in negligent land use practices and elevated nutrient loading to the lake. This has culminated in lake eutrophication and the deterioration of water quality, especially in inshore waters, which has been linked to concomitant increases in water bourne diseases such as malaria and bilharzia. One other devastating legacy has been the proliferation of HIV/AIDS in the fishing communities; with up to 90% of some fishing communities infected. Despite the economical wealth created by the fishery, it has resulted in social disruption within the local communities, with all levels of society wishing to share in the equity from the fishery.

Box 3. Positive and negative impacts of the introduction of rainbow trout *Oncorhynchus mykiss* (Walbaum).

Rainbow trout has been introduced across the world since the mid 1800s for aquaculture and sport fishery purposes (Cowx, 1997). Aquaculture production for both the table and for stocking, mainly into put-and-take fisheries, has progressively increased from around 173,000 t (worth in excess of US$500 million) in 1985 to nearly 460,000 t (worth US$1651 million) in 2005 (Fig. 2). This is mainly from outside its native range in North America, especially in Europe, Australasia and South America, where suitable climatic conditions for its production prevail. The species also contributes to the multi-billion dollar recreational fishery sector worldwide (Cooke & Cowx, 2006), These include small scale river fisheries in remote diverse places, such as the rivers of Mount Kenya and Pakistan, to lucrative put-and take fisheries in the industrialised world (Arlinghaus *et al.*, 2002). Both the aquaculture development and proliferation of fishing opportunities have led to employment and revenue generation, especially is rural areas.

These benefits must however be balanced against the loss of biodiversity, with several species known to have become extinct due to direct predation from rainbow trout and possibly disruption of ecosystem functioning (Cowx, 1997). Unfortunately, no economic value has been placed on this loss.

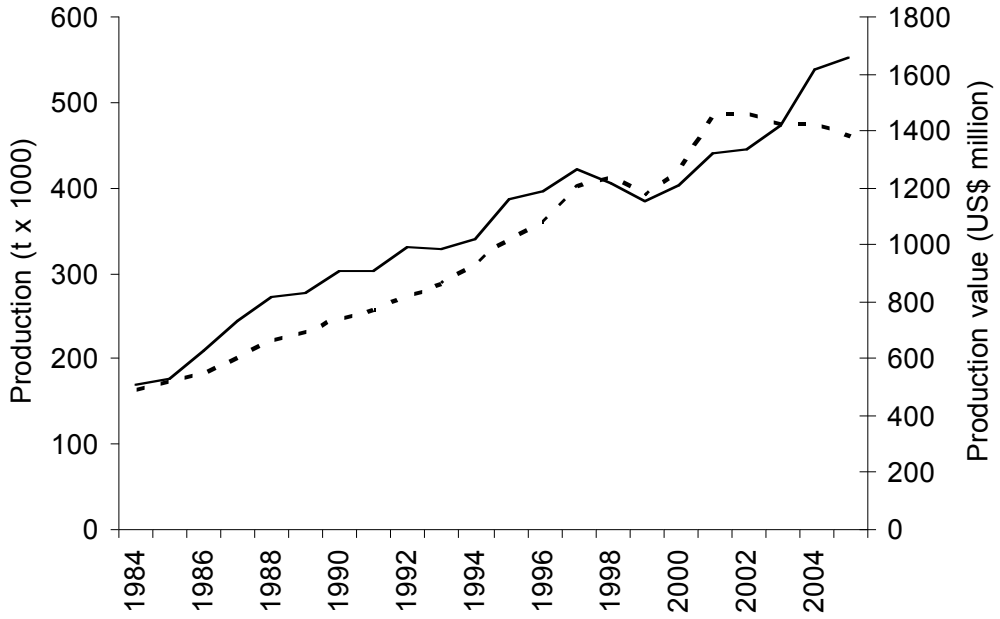


Figure 3. Volume (----; t) and value (——; US$) of aquaculture production for rainbow trout in countries outside its natural distribution range.

5- HOW ARE RISKS ASSESSED?

5.1 Risk analysis and available tools

The threats posed by non-native species to inland water species has gained increasing recognition since the 1970s (ICES, 1973; Ross, 1991; Wheeler, 1991; Claudi & Leach, 1999), not only for their environmental impacts, but also their economic and social consequences (Ciruna *et al.*, 2004b). To justify measures to mitigate threats to biodiversity (CBD, 2001), it became clear that protocols were needed to assess the risks of species introductions. These threats fall into the general area of environmental risk analysis, which has its origins in the nuclear energy industry (Cardwell, 1989), for which protocols were developed to identify, assess and manage the risks to human health of exposure to radioactivity and chemicals (Asante-Duah, 1998; Hester & Harrison, 1998). These protocols were adapted to assess the risks posed by water-borne infectious diseases (Fewtrell & Bartram, 2001) and subsequently those associated with introduced species (Calow, 1998), in particular non-native plants and their pests (Tucker & Richardson, 1995; Panetta *et al.*, 2001). Only in the last few decades have these protocols been adapted to non-native fishes, and these were mainly qualitative, with semi-quantitative elements (Kohler & Stanley, 1984; US_ANS_Task_Force, 1996; Kahn *et al.*, 1999).

Some of the more recent approaches to hazard analysis are distinctly quantitative (Kolar & Lodge, 2002), involving the use of decision-tree (CART) analysis of ecological and biological characteristics of existing non-native species in the Great Lakes region of North America to predict future invasive species from the same (Ponto-Caspian) donor region. The CART analysis approach has been trialled for non-native fishes in the UK, but is still under development (R.E. Gozlan & G.H. Copp, unpublished). Other recent approaches (US_ANS_Task_Force, 1996; Copp *et al.*, 2005b), employ multi-step frameworks, which go beyond the first phase of risk strategies (i.e. hazard identification). For example, schemes developed in the UK specifically for freshwater fishes (Copp *et al.*, 2005b), and then more generally for all plants and animals (Baker *et al.*, 2008), combine the quantitative decision-making tools required under the World Trade Organisation Sanitary and Phytosanitary (SPS) Agreement (www.wto.org) and the qualitative decision-support systems espoused by the guidelines of international policy and principles on alien species (e.g. Convention

on Biological Diversity, CoP6 Decision VI/23, 2002). To this end, protocols already in use with plants and plant pests were adapted to create a two-phase sequence of protocols for non-native freshwater fishes (Copp *et al*., 2005b). For the initial (hazard identification) phase, the *Weed Risk Assessment*, WRA (Pheloung *et al*., 1999), was converted into a Fish Invasive-ness Scoring Kit (FISK; Copp *et al*. 2008, see also http://www.cefas.co.uk/4200.aspx). Species identified by FISK as potentially invasive would then be subjected to an adapted form of the European Plant Protection Organisation (EPPO, 2000) scheme to assess the ecological, social and economic risks posed by the species.

In these non-native species risk analyses, the underlying premise is that only adverse impacts are assessed, and where information is scarce or lacking, the precautionary approach is applied (FAO, 1995). Introductions of non-native fishes, as with many other non-native plant and animal introductions, can have positive (beneficial) impacts (Gozlan, 2008), which are taken into consideration by environmental managers as part of the decision-making process. Although the WRA, and indeed all of these hazard identification systems, are not 100% reliable (Smith *et al*., 1999; Gordon *et al*., 2008), they were developed using the precautionary approach and provide simple, bibliographic-based, objective tools that facilitate the decision-making process and also help identify gaps in knowledge (Copp *et al*., 2009).

The invasive species environmental impact assessment (ISEIA) scheme introduced in Belgium is a screening system to aid in the management decision-making process. This scheme is illustrated in a two-dimensional matrix (Fig. 4), which cross-references an impact assessment outcome against the stage of invasion with the intention of identifying those species of most concern and deserving of management action. The ISEIA scheme is discussed further in Section 6, and details on how it is used in Belgium can be found at: http://ias.biodiversity.be/ias/definitions#harmonia.

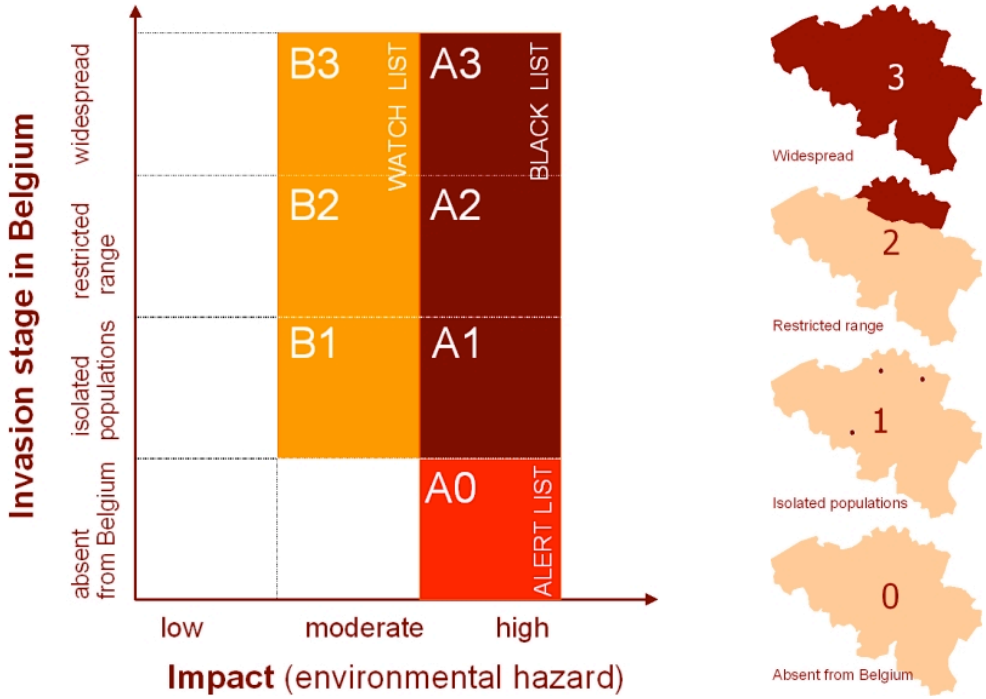


Figure 4. Assessment scheme (ISEIA) used to categorise introduced non-native species in Belgium on the basis of their impact and stage of invasion. (http://ias.biodiversity.be/ias/definitions#harmonia).

As part of a horizon-scanning process to identified future invaders, the Belgium ISEIA has recently been adapted in England to concentrate on species not yet present in England and those present but either not established or established in a restricted area (Parrott *et al.*, 2009). The outcome of the adapted ISEIA scheme, which is based on a total of 14 questions, six of which consider the species' introduction pathway(s) as well as its invasion status and eight of which concern the species potential impacts on native organisms and the receiving ecosystems. For freshwater fishes, the adapted ISEIA scheme under-estimated the potential risk of the freshwater fishes, relative to the ranking provided by FISK (Copp *et al*. 2009), which is based on a more comprehensive assessment (49 questions) that encompasses and exceeds the questions included in the ISEIA scheme. Of the calibrated FISK scores for 12 fish species tested (Table 3), eight fall within the 'high risk' category (Copp *et al*. 2009). Using the adapted ISEIA scheme, all but four species are classed as 'low risk', whereas most of the FISK scores range from the middle of the medium risk category to the lower third of the 'high risk' category (Table 3).

Table 3. Comparison of invasiveness risk scores for fresh and brackish water fishes using the adapted ISEIA scheme (Parrott *et al*., 2009) and the FISK screening tool (Copp *et al.*, 2009) whereby H = high risk, M = medium risk, L = low risk, with lower and middle ranks of FISK referring to a score's relative position (lower or intermediate 1/3) within that risk rank.

Latin name	Common name	ISEIA scheme	FISK invasiveness risk	
			mean score	rank
Acipenser ruthenus	Sterlet	L	16.0	lower H
Ameiurus melas	Black bullhead	M	28.8	middle H
Aristichthys nobilis	Bighead carp	L	24.3	lower H
Catostomus commersoni	White sucker	L	23.0	lower H
Ctenopharyngodon idella	Grass carp	L	24.0	lower H
Cyprinella lutrensis	Red shiner	L	18.0	lower H
Gambusia holbrooki	Eastern mosquitofish	M	21.0	lower H
Hypophthalmichthys molitrix	Silver carp	L	22.8	lower H
Misgurnus fossilis	Weatherfish	L	12.5	lower H
Neogobius melanostomus	Round goby	H	29.5	middle H
Pimephales promelas	Fathead minnow	L	19.0	lower H
Proterorhinus marmoratus	Tubenose goby	H	18.5	lower H

The reason for the underestimation of risk using the adapted ISEIA scheme is likely related to the relatively limited number of questions. Unlike FISK, which has been calibrated and validated against independent assessments of invasive status (Copp *et al*., 2009), the sample size of interrogation about the species in the adapted ISEIA scheme appears to insufficient. Adequate interrogation is an important consideration in the risk screening process, and must be balanced against the available financial resources. In other words, a reduction in the number of questions may reduce the time (and cost) needed to carry out an assessment, but the accuracy with which the assessment scheme identifies potential invaders may be compromised. In the adaptation of FISK from its parent screening tool, the Weed Risk Assessment scoring system (Pheloung *et al*. 1999), a reduction in the original number of questions (49) was considered but rejected over concerns related to an anticipated decline in accuracy. This would appear to have been a judicious decision given the outcomes of the ISEIA and FISK assessments (Table 3).

5.2 The precautionary approach – what is it and is it working?

Originally referred to incorrectly as the 'precautionary principle', the 'precautionary approach' was introduced in the 1970s (Carlberg & Evans, 1997). Although not specifically mentioned, these principles were the backbone of the first Code of Practice (CoP) on the Introductions and Transfers of Marine Organisms published by

the *International Council for Exploration of the Seas* (ICES, 1973). The 'principal of precautionary action' was adopted in 1987 by all North Sea states at the International Conference on the Protection of the North Sea, London, and subsequently incorporated into North Sea and Baltic fisheries management (HELCOM, 1992; OSPAR, 1992). Broader application of the precautionary approach came with the signing at 1992 Earth Summit in Rio de Janeiro of the Convention on Biological Diversity: to protect biodiversity, the 'lack of full scientific certainty should not be used as a reason for postponing measures to avoid or minimize such a threat' (CBD, 1992). The precautionary approach is now the corner stone of a variety of CoPs for responsible fisheries management (FAO, 1995; ICES, 1995; FAO, 1996; Shine *et al.*, 2000; ICES, 2004), which emphasize the role of predictive risk analysis to avoid or mitigate impacts.

As a consequence, the precautionary approach has become the default position in the assessment and management of non-native species (Heikkila, 2006), including those that are genetically engineered (Muir & Howard, 2004). However, the CBD's (2001) interpretation of the Rio Declaration emphasizes that the precautionary approach 'is not part of Risk Assessment. It comes within the decision making process of Risk Management'. This distinction is not necessarily made in a plethora of papers, reports, and policy documents espousing this 'better safe than sorry' strategy (e.g. Cambray, 2003; Simberloff, 2003; McDowall, 2004; Muir & Howard, 2004; Webb, 2006). For example, the ICES CoPs include a statement to emphasize that 'The precautionary principle will be taken into account in the final outcome of the risk assessment' (ICES, 2005). However, it is not clear whether this refers to the final 'risk summation' part of the risk assessment or to the subsequent 'decision-making' processes. Similarly, the precautionary approach is invoked in various risk assessment tools as a means of taking into account the absence or scarcity of reliable information on the biology of the organisms being assessed (Bomford, 2003; Webb, 2006; Donnelley, 2007). For example, the WRA (Pheloung *et al.*, 1999) and its derivatives (Copp *et al.*, 2005b; Copp *et al.*, 2009) have some scoring elements that accord higher scores when the quality of data is poor.

As a consequence, the precautionary approach has been integrated into numerous disciplines associated with risk analysis and management beyond the marine fisheries area. The effectiveness of the precautionary approach has been addressed directly in relatively few areas, and these are related to fisheries management or food safety. In fisheries management, appreciation of the precautionary approach is equivocal but generally positive (Essington, 2001; Piet & Rice, 2004; Aprahamian *et al.*, 2006), whereas in food safety these principles are not viewed as particularly helpful (Hanekamp *et al.*, 2003). There does not appear to be any published studies specifically assessing the effectiveness of the precautionary approach as regards non-native species, but some information can be gleaned from related studies. For example, Copp *et al.* (2007) reported a continuing exponential increase in the number of introduced non-native freshwater fishes to the UK, but they also observed a decrease in the rate of establishment success after the 1970s. This coincided with the development and enactment of new legislation, the *Import of Live Fish Act* (ILFA) in 1980. ILFA was intended to be precautionary, but in practice it is reactionary, as a species must be demonstrated to be likely to cause harm to be placed on the list of controlled species (Copp *et al.*, 2005a). The progressive implementation of ILFA during the 1980s led to a strengthening of controls on which types of species were imported to the UK, with temperate species (i.e. those most likely to establish) being listed and as such subject to a licensing procedure. In many cases, this additional administration discouraged importers and retailers (the latter being required to hold a licence to keep the regulated species) from dealing in the 'ILFA listed' species. As such, the decrease in establishment success of non-native fishes could be interpreted to be a consequence of the more precautionary approach taken by the UK government through the ILFA legislation. Alternatively, the decreased rate in successful establishments may reflect changes in both the intention and selection of imported species. Prior to the 1980s, fish introductions were often with the intention of establishing the species in UK inland waters, with species selected for their potential suitability to the UK climate. Whereas, in recent decades there has been a move towards more exotic species, especially in the ornamental and pet trade, and these species were less likely to establish. Similar patterns of continuously increasing imports and decreasing establishment success since the 1970s have been reported at the global scale (Ruesink, 2005), coinciding with the progressive integration of precautionary principles into international and national-level legislation. These studies

suggest that the precautionary approach is having a favourable influence, but more detailed studies are needed in this area.

6- MANAGEMENT OF INTRODUCED NON-NATIVE FISHES: REMEDIATION AND MITIGATION

6.1 Introduction

Effective management of non-native fishes begins with introduction prevention, using tools and methods already outlined in Section 5, as this optimise the potential to minimise adverse impacts and the associated costs (Myers *et al.*, 1998; Myers *et al.*, 2000; Simberloff, 2002). It is also complementary to the 'Precautionary principle' (Sandin, 1999; Foster *et al.*, 2000). Although managers might intuitively be expected to choose the prevention of introductions as their preferred form of intervention, Finnoff *et al.* (2007) argued that instead, managers frequently wait until non-native species have been introduced and only then act to limit their impact (Leung *et al.*, 2002; Carlton & Ruiz, 2005). This counter-intuitive action relates to a perception that post-introduction management is a safer choice than prevention because its productivity is relatively less risky, i.e. even with strict and expensive prevention protocols in place, introductions still often occur.

Introduction prevention must, therefore, be considered only as the starting point of the management of non-native fishes. Some introductions are intentional and make use of species that are predicted to produce substantial economic returns and societal benefits (Gozlan, 2008), for example, the grass carp was intentionally introduced into the UK as a bio-control agent to reduce macrophyte growth in certain waters (Stott, 1977). However, accidental introductions do occur and as these are less likely to have been subject to risk assessment, may be most damaging ecologically. Such introductions occur because of, for example, breaching of biosecurity at aquaculture or research facilities, and through the introduction of a contaminant or hitch-hiking species (Copp *et al.*, 1993; Copp, 2006).

Following the unregulated introduction of a non-native fish, management interventions, using appropriate remediation and mitigation tools, are required if the species' ability to establish and cause detrimental impacts is to be minimised or eliminated (Lodge *et al.*, 2006). Remediation is the process whereby the introduced

species is completely removed from the recipient ecosystem, such as through eradication, whereas mitigation concedes that either the introduced species cannot be eradicated or its impact is insufficient to warrant such an approach. As such, mitigation aims to control and contain the introduction so that the provision of, for example, ecosystem services can be maintained in the presence of the new species. The first step of any remediation and mitigation approach is to detect the species, and then there is a requirement to determine the level of risk that species poses to the recipient water body, the wider environment and socio-economic parameters. This risk-based management approach is necessary (Andersen *et al.*, 2004) because:

- Non-native fishes differ in their likelihood of becoming invasive and incurring ecological impact;
- Recipient ecological communities differ in their vulnerability to invasion and the values society attaches to them;
- Non-native fishes and their recipient ecosystems differ in their susceptibility to prevention and control; and
- There is a requirement for scarce resources to be allocated among existing and potential invaders through a process of balancing disparate risks, costs, and benefits that are not uniformly distributed under conditions of scientific uncertainty.

Consequently, risk management decisions on introduced non-native fishes cannot consider scientific evidence in isolation, but should also use legal, economic, administrative, social and cultural factors to determine an appropriate, dispassionate and rational response (Andersen *et al*. 2004). Typical remediation and mitigation responses (as summarized in Zavaleta *et al.*, 2001; Simberloff, 2002; Britton *et al*., 2008) include:

- Eradication: the complete elimination of the non-native fish population whereby only re-introduction could allow their return;
- Crisis management: no action is taken on that species until the development of a major pest issue at which point major management interventions are initiated; and

- Sustained control: the population is routinely suppressed by regular management actions.

However, a further option is 'do-nothing'. This may be considered a justifiable default option when assessment demonstrates the likely impact of the species will be socially acceptable, the cost of control exceeds the value of their impacts and/ or there are no existing management techniques available.

6.2 Eradication

In the management of non-native fishes, it has to be recognised that there is an almost total lack of extant techniques that can adequately control their distribution and dispersal in fluvial systems. Effective methods in more closed systems are generally lethal in nature, designed to incur maximum mortality rates in the target fishes, for example, the application of biocides, such as rotenone (Meadows, 1973; Ling, 2002; Allen *et al.*, 2006; Britton & Brazier, 2006; Rayner & Creese, 2006). As these techniques are generally non-host specific, collateral damage in non-target species may be considered inevitable (Meadows, 1973; Rayner & Creese, 2006) and consequently, some sections of society may find their use ethically questionable (Barr *et al.*, 2002; Sheail, 2003; Philip, 2005; Fraser *et al.*, 2006; Bremner & Park, 2007). Thus, the management response to the detection of a recently introduced non-native fish must initially aim to prevent dispersal into water courses and when a non-specific lethal method is to be used, such as biocide application, this has to be justified scientifically and in relation to the long-term ecological and socio-economic benefits that will be delivered by its application (Zavaleta, 2002; Fraser *et al.*, 2006). Even where non-lethal management method(s) are preferred, such as the physical removal of individual fish from the water body concerned (e.g. by electric fishing), legal constraints usually prevent their translocation to a new water body, and so the fish must still be euthanized.

The eradication of introduced species is viewed by many as a controversial and almost impossible goal due to its high expense and difficulty of success, as well as the likelihood of damage to non-target species (Myers *et al*., 1998; Simberloff, 2002; Britton et al. 2008). Although eradication has been used effectively against populations of invasive species, the chance of success is usually directly proportional

to the spatial extent of dispersal (Culver & Kuris, 2000; Anderson, 2005; Rayner & Creese, 2006), with eradication of fishes most effective in relatively small, closed, sparsely-vegetated water bodies, particularly when rotenone is used (Lozano-Vilano *et al*., 2006; Britton, In press). This is supported by Rayner & Creese (2006) in their review of non-native fish eradications in Australia where they discussed a series of successful rotenone-based operations including the removal of isolated populations of Eastern mosquitofish near Alice Springs, eradication of jewel cichlids *Hemichromis bimaculatus* Gill from drainage channels in the Northern Territory, and elimination of an abundant tilapia population *Oreochromis mossambicus* (Peters) from a lake in Queensland. Eradication success has also been reported from five operations to eliminate populations of topmouth gudgeon from fishing ponds in England between March 2005 and 2007 (Britton & Brazier, 2006; Britton *et al*., 2008). The largest of these water bodies was only 2 ha and all were < 5 m deep; to date, no topmouth gudgeon have been recorded since the operations were completed.

These examples illustrate that biocide application, particularly the use of rotenone, is an important and effective technique to eradicate introduced fishes (Ling, 2002). This naturally occurring ketone ($C_{23}H_{22}O_6$) works by inhibiting oxygen utilisation at the cellular level (Lockett, 1998). It is usually derived from the roots of *Derris elliptica*, a South-east Asian leguminous plant grown (Meadows, 1973). Its toxicity is variable according to species. For example, an initial dose of 2 mg L^{-1} of a formulation containing 5 % rotenone is sufficient to incur total mortality in roach *Rutilus rutilus* (L.), rudd *Scardinius erythrophthalmus* (L.), perch *Perca fluviatilis* L. and gudgeon *Gobio gobio* (L.) populations in enclosed waters; if only perch are present then the dosage can be halved (Meadows, 1973). Common carp may be killed within 20 h at 2.0 mg L^{-1} at 11ºC, whereas crucian carp are highly resistant, requiring a dosage of at least 8.0 mg L^{-1}. Similarly, topmouth gudgeon require longer periods of exposure at conventional doses (1.25-1.5 mg L^{-1}) and than most other cyprinid species tested (Allen *et al.*, 2006). Rotenone toxicity degrades following its application to a water body and this is influenced by temperature, light exposure, binding with suspended matter and absorption by bottom deposits (Meadows, 1973). Its effectiveness can also be impacted by the degree of site enclosure, for example open versus closed water bodies (Lintermans, 2000); physical water conditions, including depth, pH, and

discharge; availability of in-stream refuges, e.g. root masses, undercut banks, ground water recharge; rotenone type, e.g. liquid versus powdered rotenone, the varying active constituent concentrations (Willis & Ling, 2000); and the application method(s) used, e.g. manual, sprayed from ground or air, outboard motor mixing or dripped. The application must achieve an even distribution of the required dosage to all parts of the water body (Britton & Brazier, 2006); any deep holes or springs will need particular attention (Meadows, 1973) and areas of extensive macrophyte growth may provide refuge for juvenile fishes as they impede rotenone penetration (Almquist, 1959).

A major issue with rotenone application is that it is non-host specific, so unless preventative steps are taken, collateral damage in non-target species will be incurred. Although non-target fishes can be physically removed from the water and held off site until the conclusion of the project (Britton & Brazier, 2006), high mortality of aquatic invertebrates may be considered largely unavoidable and this may hinder subsequent attempts to restore the native fish community (Rayner & Creese, 2006). To overcome such issues, alternative methods of rotenone application have been developed, including oral application using baits containing a lethal dose, although Gehrke (2003) demonstrated that rotenone baits were successful in killing target species (common carp), high mortality was also observed in non-target species as they also took the baits.

When an introduced non-native fish is detected in a small lake/pond and eradication is the preferred management option, then an alternative method is to drain and disinfect the water body (Britton *et al*., 2008). Where this is considered, it is important that the hydrology of the lake basin is studied carefully as underground springs, land drainage and seepage can cause rapid refilling. Although the draining of a small pond may appear relatively simple to achieve, it can be problematic in reality. If the water is being discharged onto surrounding land that it is already saturated from heavy rain or has geology consisting of largely impermeable material, then the water will run-off and eventually reach land drains and channels. Thus, unless steps are taken, there is a risk of inadvertently dispersing the non-native fish into river catchments. Once fish have been removed, unless they can be translocated to an alternative water body (unlikely due to the presence of the non-native fish), they must be euthanized.

6.3 Control and containment

Where eradication is a remediation tool that aims to eliminate the population completely, mitigation tools, such as use of control and containment techniques, aim to suppress the introduced fish and facilitate the continued provision of ecosystem services whilst generally acknowledging that eradication may be impossible to achieve. For example, whilst pikeperch *Sander lucioperca* (L.) in England may be impossible to eradicate because of its wide distribution in river systems (Davies, 2004), this does not mean that control and containment effort should not aim to prevent its spread into 'clean' catchments, using existing legislation (Hickley & Chare, 2004).

In the control and containment of non-native fish populations, small-scale eradications can be integrated into the approach whereby populations are eliminated from waters from which there is a high chance of their dispersal into fluvial environments (Britton *et al*., 2008). In these situations, biocide application and drain-down and disinfection methodologies may be used accordingly. However, other management options exist. The physical removal of the introduced species from infected waters using methods such as seine netting, electric fishing and gill netting reduces their abundance in the site and enables all non-target species to be returned. Knapp & Matthews (1998) reported on an eradication exercise to manage invasive brook trout *Salvelinus fontinalis* (Mitchill) from mountain lakes in Sierra Nevada, California, USA, where they estimated that up to 20 % of the lakes were suitable for gill netting as a viable alternative to rotenone application. Similar findings were reported by Neilson *et al*. (2004) following their use of physical removal methods for controlling invasive rudd in ponds in New Zealand.

An alternative method is biocontrol, which relies upon natural enemies to attack the introduced non-native species (Secord, 2003). Classical biocontrol introduces a natural enemy from the natural range of the species to control it in its new environment, whereas augmentative biocontrol enhances populations of native predators, parasites and/or pathogens to improve their regulation of the introduced species (Secord, 2003). The aim is to take advantage of negative species interactions to reduce the survivorship and/ or reproduction of the introduced species. It is rarely

capable of eliminating the target species, but aims to keep it suppressed at socio-economic and/ or ecologically acceptable levels, with only a minimal input of effort (Gause, 1969). The most successful biocontrol agents are those that not only damage its target, but does so in a host-specific way so that collateral damage is minimised (Secord, 2003), as they can then be considered environmentally 'safe and clean' (Doutt, 1971; Simberloff & Stiling, 1996a; b). However, unlike rotenone application, their ecological effects are likely to be irreversible; once an introduced biological agent has established, only eradication may remove it.

Although the eradication and control methods outlined have the potential to manage the dispersal of introduced non-native fishes, all have controversial elements. To overcome some of the more contentious aspects of these methods, novel techniques have been/are being developed and tested; when adopted within integrated management programmes for non-native fish control, they have the potential to provide, for example, enhanced capture rates and increased suppression of pest populations. For example, to control introduced species that have predominantly genetic, but environmentally reversible, sex determination, such as many fishes, models have been developed and tested that use carriers of Trojan Y chromosomes (individuals that are phenotypically sex reversed from their genotype) (Gutierrez & Teem, 2006; Cotton & Wedekind, 2007). They demonstrate that repeated introduction of YY females into wild populations may produce extreme male-biased sex ratios and the eventual elimination of XX females, thus leading to population extinction in the target species (Gutierrez & Teem, 2006). However, there are a series of issues that remain unresolved, for example little is known about the relative performance of genetically manipulated individuals in the wild, or about the sex determining systems of many invasive species (Cotton & Wedekind, 2007). Introductions of inducible fatality genes have also been mooted (neutral genes that become lethal when activated by an external agent or stimulus) but are problematic in that they must become fixed or achieve high frequencies in the population before activation (Muir & Howard, 2004).

A more practical technique may involve pheromone use (Burnard *et al.*, 2008). Most fishes rely on pheromones (chemical signals released by conspecifics) to mediate social behaviours, with three categories discerned based on their function: anti-

predator cues, social cues, and reproductive cues (Sorensen & Stacey, 2004; Burnard *et al.*, 2008). Each of these categories comprises pheromones that can induce 'primer' effects (developmental and/or endocrinological changes) and/or 'releaser' effects (strong behavioural changes). Those fish pheromones that have been chemically identified are very potent; in combination with their specificity, this provides them with considerable potential for use in controlling introduced non-native fishes (Burnard *et al.*, 2008). Already successfully used within integrated control programmes for sea lamprey *Petromyzon marinus* in the North American Great Lakes (Jones, 2003), pheromone cues are used simultaneously to exploit multiple weaknesses in the life history of the species whilst recognising stock-recruitment relationships. Sorensen & Stacey (2004) recommended that their use involves a variety of pheromones to supplement and increase the efficiencies of other control strategies, for example to:

- Facilitate trapping efficiency (Twohey *et al.*, 2003a; Twohey *et al.*, 2003b);
- Disrupt/ reduce reproductive success (Carde, 1997; Wyatt, 2003);
- Disrupt movement and migrations (Li *et al.*, 2003; Sorensen & Vrieze, 2003); and
- Promote the success of sterilised fish and to repel others from sensitive areas (Maniak *et al.*, 2000).

Control and containment can also be achieved through the construction of barriers to prevent the upstream movement of an invading species. Examples of physical barriers include the construction of low head barriers to prevent sea lamprey from reaching spawning grounds in the Great Lakes system (Great Lakes Fishery Commission 2009). These block the passage of lampreys, but allow desirable fishes to pass. Their location downstream of spawning grounds prevents access to suitable spawning habitat, eliminating larval lamprey production and the subsequent need for lampricide treatment. Electric and hydroacoustic barriers have also been used to prevent the movement of invading fishes. Experiments in the USA revealed that a cross-channel a air bubble curtain barrier containing pneumatically generated sound signals randomly selected from a predetermined frequency range was 95 % successful in repelling the movements of bighead carp *Hypophthalmichthys nobilis* (Taylor *et al*. 2005).

6.4 Rapid detection, assessment and response to introduced non-native fishes

Introductions of non-native species are generally easier and cheaper to control and/or eradicate when their distribution is still highly localised (IUCN, 2000; Manchester & Bullock, 2000; Genovesi, 2005; Cacho *et al.*, 2006). This has given rise to the development of the three-step management concept of 'Rapid detection, rapid assessment, rapid response' (Myers *et al.*, 2000; Zavaleta *et al.*, 2001; Zavaleta, 2002; Anderson, 2005). A major problem with the concept is the ability to detect a newly-introduced species is compromised by its limited distribution, as the effort required for detection is usually inversely proportional to population size (Hayes *et al.*, 2005). Hence, networks designed to detect new introductions rapidly must maximise their effectiveness by emphasising their work on high-priority targets, such as high-risk locations, high-value resources, important pathways, and populations and species of most specific concern (Lodge *et al.*, 2006).

When a non-native fish is detected soon after introduction and has yet to establish a sustainable population and disperse, then cost-effective options are likely to still be available; if eradication is identified as an appropriate response, then it carries relatively little risk due to the chemical or mechanical control efforts remaining highly localised (Genovesi, 2005). If, however, the introduced species has already established population, then options for management are already being constrained, particularly if that species is now spreading. Its control or eradication is now more difficult and expensive due to the increased spatial and temporal extent of application, and there is a corresponding increase in the risk of failure (Genovesi, 2005; Rayner & Creese, 2006). If the introduced species is only acted upon when it is highly invasive and spreading rapidly, then realistic opportunities for control may be slim, and mitigation schemes that compensate for the presence of the species may be the only appropriate management option.

To promote the reporting of newly introduced fishes, regular engagement with key stakeholders is advisable, for example through the development of effective education schemes including the dissemination of relevant information (Fig. 5).

Following detection of a new, unregulated introduction, rapid assessment is required to assess the risks to both the receiving water and to the wider environment. The risk

management procedures used in pre-introduction analyses (Section 5) are highly appropriate for this and although being used post-introduction, remain valid in determining the level of risk posed by the introduced species. However, following the introduction, risk management must also consider the environmental, logistical and resource constraints of the species and its associated management (Cacho *et al.*, 2006; Fraser *et al.*, 2006; Section 6.5).

Figure 5. Examples of information on non-native fish distributed by authorities to help educate passive networks.

Sources: top: http://www.seagrant.umn.edu/exotics/cards.html; bottom: http://www.ornamentalfish.org/aquanautconservation/petfishbelong.php.

Once analysis has objectively assessed the associated risks of the recently introduced fish, then an appropriate management response can be determined. However, even if this suggests the introduced non-native species represents an unacceptable ecological risk, and eradication, using an appropriate method, is the most appropriate response, there may be overriding constraints that will inhibit this. For example, if rotenone application was considered as the optimum management intervention, then it may result in substantial collateral damage (Ling, 2002; Rayner & Creese, 2006; Britton *et*

al., 2008); if those species are of high conservation values, then this may be unacceptable (Dean, 2003). Thus, following an introduction, risk analysis must either be used in conjunction with, or incorporate, other decision support tools (Simberloff, 1997; Myers *et al.*, 2000; Simberloff, 2002; Zavaleta, 2002; Genovesi & Shine, 2004; Cacho *et al.*, 2006; Fraser *et al.*, 2006; Lodge *et al.*, 2006). Other issues that should be considered are:

- Resources: eradication exercises are often expensive and it is vital to determine who is to meet the economic and manpower costs.
- Opposition: even during the early stages of an invasion by an non-native species, eradication attempts remain controversial to the public, politicians and some scientists who believe it is unachievable, unrealistic and, often, morally unacceptable.
- Inadequate legislation: facilitating the rapid initiation of eradication exercises is their basis in legislation; where detection of a new introduction requires an effective response that has a basis in law, there is an increased chance of an eradication attempt being initiated quickly.
- Invasion pathway remains open: unless the pathway by which the invader arrived has been closed, for example through increasing the biosecurity of an aquaculture facility, then re-introduction of the non-native species remains a possibility.
- Feasibility of eradication: feasibility can be determined by analysis of logistics, the biological characters of the species concerned, the water characteristics and the management required to achieve eradication.
- Presence of protected species: consideration must be given over the presence in the receiving water-body of any native species with legal protection or high conservation status that would be adversely impacted by any eradication attempt.
- Resource value: if receiving water is an exploited fishery of high societal and economic value, assessment must be made of how this will be impacted by the eradication attempt.

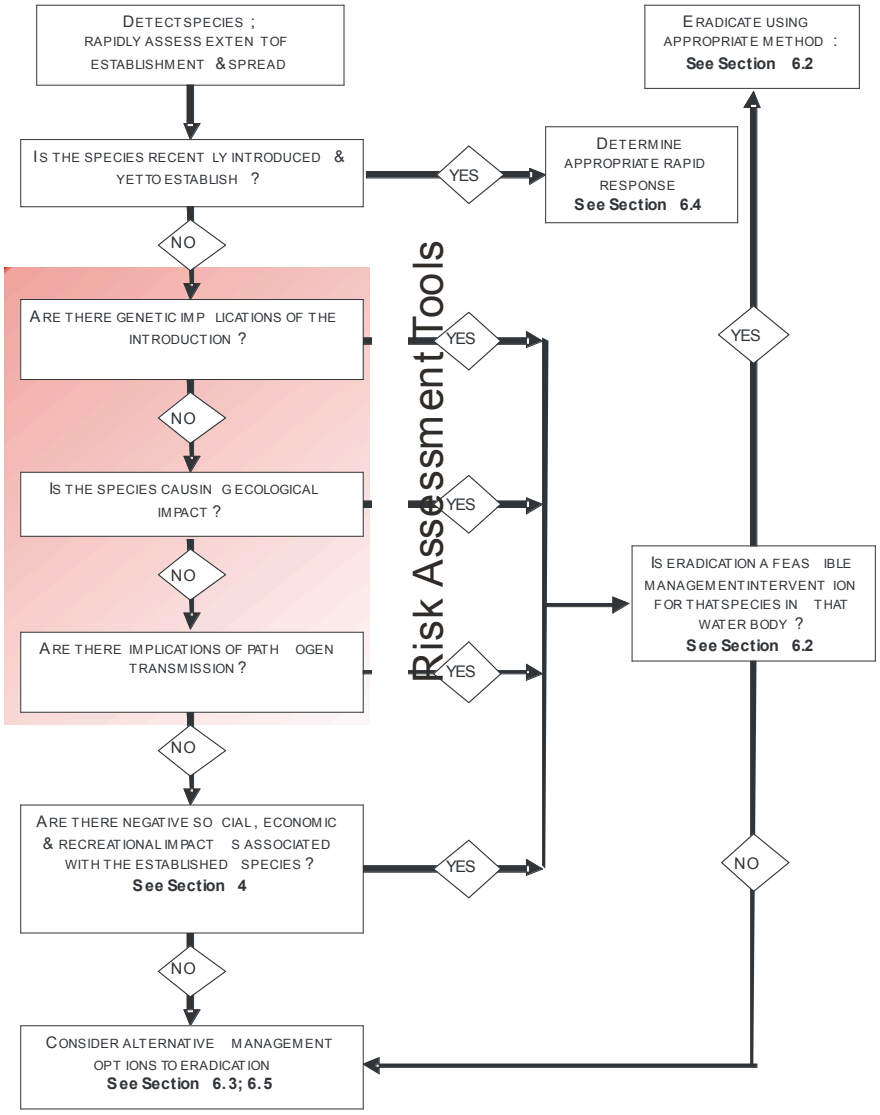
6.5 Management assessments

The determination of the most appropriate management response to an established non-native fish requires use of risk identification, risk assessment and risk management, which collectively represent risk analysis. These are interacting - but functionally separate - activities; risk identification determines whether a species is

likely to pose a threat (of establishing, dispersing or exerting an impact); risk assessment characterises the likelihood and severity of potential adverse effects of the introduced fish, whereas risk management is the process of identifying, evaluating, selecting, and implementing actions to reduce that risk (Andersen *et al.*, 2004; Section 5). The general approach to risk assessment of non-native fishes focuses mainly on problem formulation; classification schemes are developed that predict invasiveness, identifying pathways of introduction, characterising susceptible resources and predicting the ecological consequences. However, following introduction, risk must also be managed in accordance with available resources, the characteristics of the recipient ecosystem, the resident communities and species of the recipient ecosystem, and a combination of contemporary science and the public-policy choices.

The Belgian scheme presented in Figure 4 uses a two-dimensional ordination (Section 5) that compares impact against the stage of invasion. Whilst this provides assessment of those species that are most deserving of direct management intervention through the extent of their invasion, it does not guide the manager through a quantitative risk-analysis framework that also considers resource availability, feasibility of eradication/ control, measures that may be required to restore suppressed native species etc. How such criteria may be used within a risk analysis procedure is shown diagrammatically in Box 4, adapted from Copp *et al*. (2008). The entire process would also have to incorporate several other components to provide an effective overview of the whole procedure.

BOX 4: RISK ANALYSIS FLOW CHART FOR AN INTRODUCED NON -NATIVE FISH



7- CLIMATE CHANGE IMPLICATIONS FOR NON-NATIVE FISHES

7.1 Change in temperature

Mean global surface temperatures have increased by approximately 0.6 ^{o}C over the last 100 years (Mooij *et al.*, 2005), and the Intergovernmental Panel on Climate Change (IPCC) predicts a mean increase of 1.4 to 5.8 ^{o}C for 2100 unless action is taken against human-generated greenhouse effects. Irrespective of introductions of non-native fishes, the ecological influences of these temperature increases are likely to be significant for many fishes (FSBI, 2007). For example, climate-change induced increases in evapo-transpiration and reductions in precipitation are predicted to be important drivers of freshwater fish loss in many areas of the world (Xenopoulos *et al.*, 2005). Temperature is a fundamentally important abiotic factor that regulates many processes for fishes (Magnuson *et al.*, 1990), with most physiological processes heavily influenced by temperature, including spawning, ontogeny and growth (Fry 1968; Tonn, 1990). As a result of thermal niches, the distribution of fishes is largely temperature dependent, and this is well reflected in species' natural range limits according to latitude and altitude (Magnuson *et al.*, 1979).

In the northern hemisphere, the consequences of the predicted temperature increases have been typically described as shifts in community structure towards more thermophilic and southern taxa (Portner *et al.*, 2001; Soto, 2001; Daufresne *et al.*, 2004; Fang *et al.*, 2004; Jansen & Hesslein, 2004). This implies that the biological range of species that are close to their thermal limits will increasingly move north in alliance with geographic shifts in their preferred temperature range, with the exception of species confined to enclosed waters from which migration is not possible (Dembski *et al.*, 2006). For example, increasing water temperatures have already been identified as likely to result in the increased colonisation and range expansion of a number of fishes in North America (Casselman, 2002; Jackson, 2002; Magnuson, 2002; Vander Zanden *et al.*, 2004; Sharma *et al.*, 2007). In Canadian fresh waters, it is anticipated that species such as the smallmouth bass *Micropterus dolomieu* Lacepède will increasingly colonise new habitats further north as the amount of suitable thermal habitat increases (Magnuson *et al.*, 1997; Casselman, 2002; Sharma *et al.*, 2007). Some fish species are expected to expand their northern range by approximately 500 to 600 km (Magnuson *et al.*, 1997; Jackson, 2002; Rahel, 2002; Vander Zanden *et al.*,

2004). Given the ecological sensitivity of some of these northern habitats, this may have serious consequences for many indigenous fishes (Casselman, 2002; Magnuson, 2002). A futher concern is that these habitats will become more susceptible to colonisation by non-native fishes being transported in ballast waters from Eurasia (Magnuson *et al.*, 1990).

An extremely important implication of climate change concerns those habitats that have already been subject to introductions of non-native fishes that are still within their lag phase, i.e. the species has yet to establish/become invasive. The ability of a non-native fish to become invasive following introduction is dependent upon various environmental and biological factors, including the level of inherent biotic resistance and resilience in the receiving ecosystem and the ability of individuals to be able to reproduce in their new environment (Shurin, 2000). The duration of the lag phase of an introduced fish can be very long (decades, or longer) and is determined by factors including inherent lags in population growth, prevailing environment parameters and delays in local adaptation (Crooks & Soule, 1999; Facon *et al.*, 2005; Fausch, 2007). Trigger events that may terminate a lag phase include increased temperatures that now enable successful reproduction and recruitment, and decreased biological resistance and resilience in the ecosystem arising from increased thermal stress on native species (Hessen, 1996; Petchey *et al.*, 1999). Should native communities be modified and/or destabilised by temperature increases, invasion pathways for some introduced warm-water fishes may open further, especially for those species that have already been purposely introduced into cool-water, temperate regions (e.g. for aquaculture and angling) in the belief that they could not become invasive because of temperature restrictions (Hickley & Chare, 2004). This climate-change induced invasion may involve ecological trade-offs that suppress the slow-growing, cool-water native species and favour the fast-growing, competitively superior, introduced species (Williamson *et al.*, 2001; Williamson *et al.*, 2002). It also demonstrates that species that are being considered for introduction in a prevailing temperature regime must also be assessed under conditions that are likely to be encountered in future under a range of climate change scenarios (Section 5).

7.2 Change in rainfall

In many temperate regions, the increase in temperature is predicted to be allied to changes in rainfall patterns, with low rainfall in summer, increased precipitation in winter and the increased incidence of extreme climatic events such as large storms in summer. As this suggests an increased intensity of flood events, this presents an indirect potential threat of the increased dispersal of non-native fishes from lentic into lotic habitats in these high water periods. For example, in England, flooding of both aquaculture sites and lakes in the floodplain has already resulted in species including European catfish, white sucker *Catostomus commersoni* Lacepède and topmouth gudgeon entering river catchments, so opening potential pathways of invasion (Copp *et al.*, 1993; Britton & Brazier, 2006; Britton *et al.*, 2007). Similarly, extreme flood events elsewhere have also resulted in the escape of non-native fishes into the wild from aquaculture sites, for example the common carp entered Lake Naivasha, Kenya, in 1999 following large floods arising from heavy rains associated with El Nîno (Hickley *et al.*, 2004).

7.3 The aquaculture perspective

The influence of climate change on the establishment and invasion of non-native fishes has, so far, been discussed as negative, with range expansions and increased interactions between thermally-stressed native fishes and thermophilic, competitively superior, non-native fishes. However, increased waters temperatures may also bring some socio-economic benefits, particularly in relation to aquaculture. For example, European inland aquaculture is presently dominated by trout production in Western Europe and common carp in Eastern Europe, and although pond culture continues to be an important sector, there are current climatic limitations to production (Varadi, 2001). Key to the future development of European pond aquaculture is diversification, particularly in relation to species (Varadi, 2001). With increased water temperatures, the reliance of production on a small number of trusted cool-water species may be at least partially replaced by the use of new species more commonly used in warm-water aquaculture, such as tilapias (El-Sayed & Kawanna, 2008). Species diversification may also include the use of species typically used in aquaculture in Asia, such as Indian major carps and Chinese carps (Munilkumar & Nandeesha, 2007). Where such warm-water species are already being cultured in temperate European climates, increased ambient temperatures may facilitate increased production and reduce the

input of energy required to maintain the requisite water temperatures. For example, a barramundi aquaculture site in Southern England has to maintain water temperatures at a constant 28 ^{o}C for production (Bird, 2006), whereas summer air temperatures in the region currently rarely exceed 20 ^{o}C.

These examples demonstrate that the potentially increased commercial viability of some species in aquaculture could certainly provide some significant benefits to aquaculturists and consumers in temperate regions. This should, however, be done in strict accordance with risk assessment processes and high biosecurity to minimise the risk of these species to the wider environment.

8- CONCLUSIONS AND RECOMMENDATIONS

This briefing paper focuses on the issues associated with introductions of non-native fishes and reviews current understanding of the subject by drawing on past positions and suggesting improved regulations, as well as predicting future trends. The subject of regulating the introduction and presence of non-native fishes in the wild is sensitive and difficult to review, as it falls in a vacuum between socio-economic drivers, conservation practitioners and a wide range of academic disciplines that often ignore each other in their respective conclusions. In this document, a pragmatic approach has been taken to the neo-paradox, resulting from post mid-twentieth century economic development, that, despite knowledge of potential adverse impacts on recipient ecosystems (Section 3), economic drivers are still pushing for further introductions (Section 2). For example, the European aquaculture sector is fairly limited compared with Asia and is under pressure from the European Community to expand. An option is market diversification, whereby an increased range of species are cultured. As the majority of these species are likely to originate from outside of Europe, this will increase the risk of new introductions arising through lapses in the biosecurity of aquaculture sites.

Understanding and acknowledging this paradox is crucial to regulate adverse species introductions successfully, as it enables market diversification to proceed in conjunction with robust risk assessment tools, more efficient mitigation and, ultimately, the more efficient regulation of future introductions. Pragmatically, the absolute zero risk advocated by some conservationists is an unrealistic position that to date has not been efficient in prohibiting the introduction of fish species worldwide or limiting their adverse impacts. Policy advisors are not in a position to ignore the risk, but neither are they in a position to limit trade based on the precautionary approach, as policy makers are often unable to wait for complete scientific answers prior to policy development and implementation. This briefing paper has revealed that manipulating ecosystems can never be considered risk free, and increased international regulation structured around sound risk assessment tools is a major step forward in regulating future fish introductions (Section 5). Risk assessments are developed and based on current knowledge, where level of risk is constantly updated based on current scientific understanding of the impact of non-native species on ecosystems. Limiting

and managing risk is key (Section 6) and as highlighted in Gozlan (2008), the risk is not evenly distributed across species and ecosystems (Section 3). This document argues that although mostly empirical, scientists today have a greater understanding of the risks associated with each species introduction. Some introduced species have often been associated with adverse impacts on ecosystems (e.g. black bullhead; blackchin tilapia *Sarotherodon melanotheron* Rüppell; bluegill *Lepomis macrochirus* Rafinesque) whereas others have not (e.g. Arctic char, *Salvelinus alpinus* (L.); cachama *Colossoma macropomum (Cuvier)*; Danube sturgeon *Acipenser gueldenstaedtii* Brandt & Ratzeburg). This briefing paper recommends that by combining the economic values of species on a global scale with their associated level of ecological risk, those of low ecological risk and high economic value (i.e. sterlet sturgeon *Acipenser ruthenus* L.; coho salmon *Oncorhynchus kisutch* (Walbaum); catla *Catla catla* (Hamilton)) can be teased out.

Figure 6. Relationship between economic value and likelihood of ecological impact per given species. The dashed line shows the average likelihood for all fish species introductions. White circles indicate species production in tonnes strictly below

100,000; grey circles indicate more than 100,000 and less than a million and black circles indicate strictly above a million (Gozlan 2008).

In addition to recommending the use of standardised, robust risk assessment as a basis of future fish species introductions, there is an associated need to support up-to-date research on the ecological implications of non-native species, in particular in the causal relationship between species introductions and loss of biodiversity. An understanding of the processes involved will refine the risk analysis tools as well as limit the risk of adverse effects. This is particularly relevant in the context of climate change where the equilibrium of ecosystems will be modified and consequently, the potential impact of fish introductions.

It has also emerged from this review that one the greatest risks associated with this global movement and introduction of species is the introduction of non-native infectious agents (cf. Section 3). This is where the magnitude of risk is the greatest and where monitoring and future efforts should concentrate.

Finally, maps of conservation hot spots should be drawn based on a combination of predictors of future introductions (e.g. human population density, aquaculture activity) and risk level of incurring losses to local/global biodiversity (e.g. number of critically endangered species). This would accelerate prioritisation of the areas at national and global levels where risks of future introductions should be minimised, such as those where endemic species are at risk and the intensity of introductions of non-native species has been limited.

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