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Towards the conservation of freshwater fish: Iberian Rivers as an example of threats and management practices

Alberto Maceda-Veiga

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Abstract The current freshwater fish fauna crisis is such that natural resource managers urgently need to identify priorities and understand the management consequences of actions aimed at maximizing the preservation of biodiversity. Freshwater research is often poorly linked to conservation ecology; and interdisciplinary studies illustrating examples of freshwater ecosystem conservation are scarce. The Iberian Peninsula has a long history of anthropogenic disturbance that has led to the poor conservation status of its ichthyofauna, with 52 % of species now catalogued as critically endangered, endangered or vulnerable, according to UICN criteria. This paper gives an overview of the main threats (habitat degradation, hydrological alterations and exotic species) that have altered the function and connectivity of Iberian rivers. Case-study examples are provided to analyse the repercussions of these threats and the management actions planned or already performed in these systems. The interaction of many threats is responsible for native fish decline. However, freshwater managers and researchers should not let the trees prevent them from seeing the overall wood, when seeking to achieve practical solutions with the best balanced cost benefit and the collaboration of all ecosystem researchers and stakeholders. Conservation efforts should be focused on the preservation of ecological processes, in order to achieve the goals of the Water Framework Directive and guarantee the conservation of Iberian native fish species.

Keywords Aquatic conservation · Management actions · Endangered fish · Freshwater · Anthropogenic modifications

A. Maceda-Veiga (✉)
Department of Animal Biology (Vertebrates) and Research Institute of Biodiversity (IRBio), Faculty of Biology, University of Barcelona, 08028 Barcelona, Spain
e-mail: albertomaceda@gmail.com

Introduction

Freshwater ecosystems are some of the most threatened in the world. The loss of their biodiversity appears to be more intense than that of any other habitat (Dudgeon et al. 2006; Clavero et al. 2010; Olden et al. 2010; Moyle et al. 2011). The number of endangered freshwater species on the International Union for the Conservation of Nature (IUCN) Red List has more than tripled since 2003 (Vié et al. 2009). This concern is great in regions with a high degree of endemicity like the Mediterranean area in which 70 % of freshwater species are catalogued as threatened by extinction or are already extinct (Smith and Darwall 2005). This percentage is the highest recorded anywhere in the world for any taxonomic group (Vié et al. 2009). Examples of freshwater fish decline and studies of population trends are common in the scientific

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literature (Elvira 1995a, b; Aparicio et al. 2000; Leprieur et al. 2008; Maceda-Veiga et al. 2010b; Moyle et al. 2011). Many of these studies have identified common causes of population decline (Elvira and Almodóvar 2001; Clavero et al. 2004; Nilsson et al. 2005; Dudgeon et al. 2006; Clavero et al. 2010; Gozlan et al. 2010; Olden et al. 2010; Hermoso and Clavero 2011). The poor conservation status of Iberian freshwater fish species requires the design and implementation of conservation plans (Filipe et al. 2004; Abell et al. 2007; Araguas et al. 2009; Nel et al. 2009; Hermoso and Clavero 2011; Moyle et al. 2011; Olden et al. 2011). However, examples in the scientific literature of management action have long been biased towards terrestrial ecosystems; research into freshwater ecology is still not often linked to conservation ecology. Thus, interdisciplinary practical studies illustrating examples of freshwater fish conservation are scarce (Lake et al. 2007; Nel et al. 2009; Strayer and Dudgeon 2010).

Natural resource managers need to identify priorities and understand the management consequences of actions aimed at maximizing the preservation of biodiversity (Dudgeon et al. 2006; Helfman 2007; Strayer and Dudgeon 2010; Olden et al. 2011; Grantham et al. 2012). Species decline may result from the synergy of several impacts or from the specific effect of one, whose proportional contributions to fish decline may also vary over time (Groom et al. 2006; Olden et al. 2010; Hermoso and Clavero 2011; Grantham et al. 2012). While large-scale assessment can highlight the overall extent of the freshwater crisis, severity and causes are best understood, even in large climatic areas (Hermoso and Clavero 2011), by studies that focus on the regional level (Moyle et al. 2011). The Iberian Peninsula provides a suitable scenario for analysing the decline of freshwater fish because it has a high degree of fish endemicity and a long history of anthropogenic disturbance (e.g. dams, forest exploitation, water extraction, extensive agriculture, water pollution, industrialization, extensive forest fires, exotic introductions, fisheries) (Sabater et al. 2009; Clavero et al. 2010; Hermoso et al. 2011; Maceda-Veiga and De Sostoa 2011). This paper gives an overview of the main threats to Iberian freshwater ichthyofauna. It shows the consequences that various management actions have had or may have on their conservation, using interdisciplinary case-study examples. The information provided may encourage researchers to develop practical studies and help resource managers in the conservation of freshwater biota, especially in areas with similar climatic conditions.

Methods

Sources of information

All native freshwater fish species from Iberia were considered in the study, including anadromous and catadromous species (e.g. *Anguilla, Salmon*, *Alosa*) and endemic species present in coastal lagoons (e.g. *Aphanius, Valencia*), but excluding amphidromous species like mugilids. The IUCN Red List of Threatened Species (www.iucnredlist.org) was used to compile information on the conservation status, threats, population trends and conservation tools required. This information, often used in similar studies (e.g. Clavero et al. 2010), was collected through a regional assessment of fish endemic to the Mediterranean basin by some 20 experts in fish biology and conservation (Smith and Darwall 2005). The examples used to illustrate potential or actual management actions are based on reports and papers identified via an intensive literature review, and by personal communication with freshwater managers and researchers or with ecosystem stakeholders. The main threats discussed are based on IUCN assessment and on the experience of researchers working on freshwater ecosystems, especially in regions with similar climatic conditions (Doadrio 2001; Elvira and Almodóvar 2001; Sabater 2008; Sabater et al. 2009; Clavero et al. 2010, Elosegui et al. 2011; Hermoso and Clavero 2011; Grantham et al. 2012).

Conservation status of Iberian ichthyofauna

The conservation status of many species is critically endangered, endangered or vulnerable. Populations are declining sharply, like freshwater ichthyofauna elsewhere (Moyle et al. 2011; Hermoso and Clavero 2011) (Fig. 1, Appendix 1). Freshwater fish diversity in the Iberian Peninsula is characterised by a high degree of endemicity (73 % of species are endemics to the Iberian Peninsula) and restricted distribution ranges (Fig. 2, Appendix 1). Exotic fish species

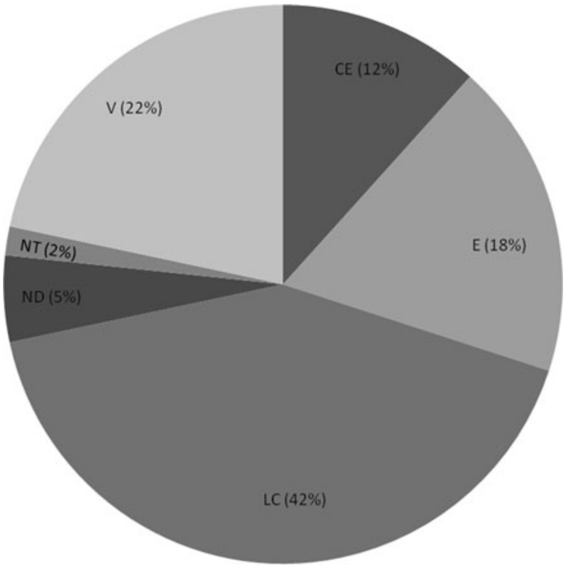

Fig. 1 Current conservation status of Iberian native fish species based on UICN criteria: *CE* critically endangered, *E* endangered, *NT* near threatened, *V* vulnerable, *LC* least concern and *ND* without enough data

account for 42 % of Iberian freshwater fish, which is similar to that reported for other European countries (Elvira 1995a, Doadrio 2001, Leprieur et al. 2008, Maceda-Veiga et al. 2010a, Appendix 2). However, the relative proportion of exotic species might be attenuated by the description of new native species (Kottelat and Freyhof 2007; Appendix 1).

Threats to freshwater biodiversity

According to UICN, the main threats to Iberian ichthyofauna are water extraction (including hydrological infrastructures) and introduced species, followed by climate constraints, pollution and overexploitation (Table 1). These threats are shared with other Mediterranean regions (Hermoso and Clavero 2011). With a view to developing appropriate management actions, the following subsections summarize how these and other reported threats to freshwater fish fauna in Iberia are affecting all key components of these freshwater ecosystems (i.e. riparian coverage, water flow level and quality, gravel bed streams and aquatic biota), together with the action of natural restrictions from the Mediterranean climate (i.e. forest fires and droughts).

Hydrological infrastructures and water extraction

Water extraction, considered one of the main threats to freshwater ecosystems worldwide, is a particular

Fig. 2 Population trends of the native freshwater fish fauna in Iberian Peninsula

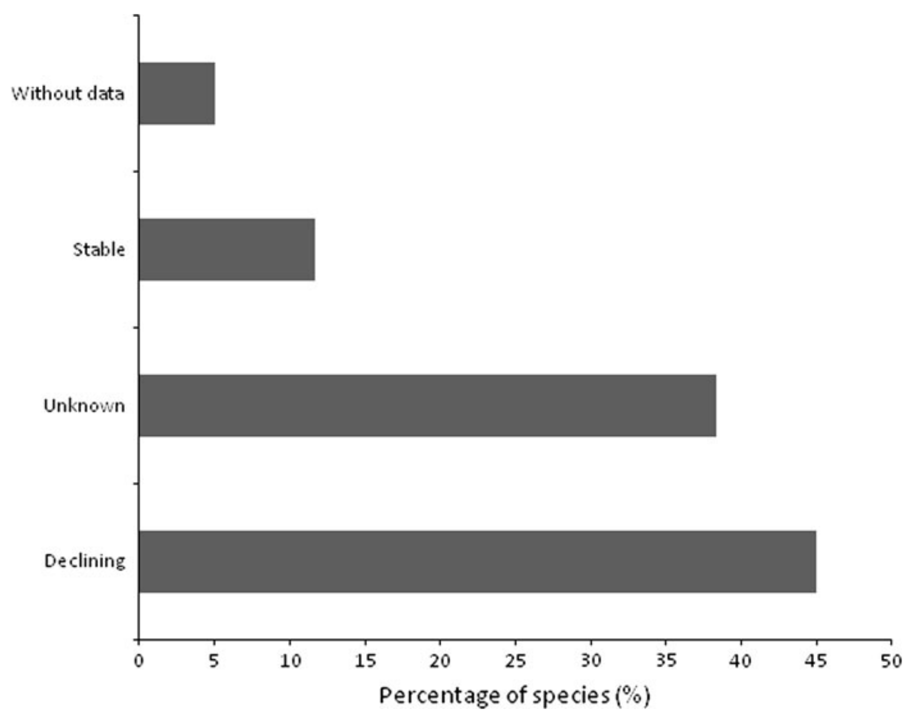

Table 1 Threats to Iberian native freshwater fish fauna and proposed management actions based on IUCN criteria

	%
Impacts	
Water extraction	60.00
Introduced species	50.00
Climate change	48.33
Pollution	21.67
Human exploitation	8.33
Urbanization	3.33
Management actions	
Habitat management	45.00
Habitat restoration	23.33
Protected areas	16.67
Species management	14.55
Ex-situ conservation	10.00

Numbers shown are the percentage of species affected

problem in arid and semi-arid regions such as the Mediterranean area (Sabater 2008; Boix et al. 2010; Clavero et al. 2010). The ratio water usage/availability ranges from 4 to 7 % in northern Iberia to 80 % in the rest of the Peninsula, increasing from northwest to southeast (i.e. from wet to arid regions) (Sabater et al. 2009). Excessive water extraction is particularly evident in the Tablas de Daimiel (Guadiana, Spain) and many rivers in the Mediterranean area (Bromley et al. 2001; Groom et al. 2006; Sabater et al. 2009). Many hydrological infrastructures were built in 1960–1990 to satisfy agricultural, hydropower and domestic demand and to assist with flood regulation (Nicola et al. 1996; Almodóvar and Nicola 1999). These physical barriers (e.g. weirs, dams, channels) cause hydrological alterations (e.g. flow regime, sediment trap), leading to irreversible alterations in river function (e.g. hypersalinization of estuaries, sudden water flow and temperature fluctuations), and directly threaten aquatic biodiversity (e.g. restriction of migration, destruction of spawning grounds, favouring invasions) (Almodóvar and Nicola 1999; Gregory et al. 2002; Rovira and Ibañez 2007; Johnson et al. 2008).

The effects of water flow reduction on fish populations are difficult to quantify because many abiotic and biotic interactions exist (Gasith and Resh 1999; Clavero et al. 2010). Small barbels (e.g. *Barbus haasi*) can survive in streams that retain isolated pools during seasonal droughts (Aparicio and De Sostoa 1999). However, a multi-year study on the effects of drought on Guadiana fish populations showed that in dry years large barbels (*Luciobarbus sclateri*) and chub (*Squalius torgalensis*) declined, but small fish species like sticklebacks (*Gasterosteus aculeatus*) were favoured (Magalhães et al. 2007). No clear pattern has been reported for exotics (Leprieur et al. 2006; Boix et al. 2010). The susceptibility of fish to disease seems to increase with supra-seasonal droughts (Maceda-Veiga et al. 2009). Indeed, low fish abundance and specimens with a deteriorating body condition were found in Sau reservoir (Catalonia, NE of Spain) after a partial reservoir drawdown to optimize water quality for domestic demand (Benejam et al. 2010).

The loss of river connectivity due to physical barriers affects long and short migratory species greatly (e.g. European eel, *Anguilla anguilla*; large barbel species, *Luciobarbus* spp.; Atlantic salmon, *Salmo salar*) and has contributed to the almost confirmed extinction of some (e.g. Atlantic sturgeon, *Acipenser sturio*) (Elvira 1995a, b; Araujo and Ramos 2000; Smith and Darwall 2005). Examples of fish species decline following damming are widespread in the literature (Almodóvar and Nicola 1999; Doadrio 2001; Gregory et al. 2002; Clavero et al. 2004). The range of the European eel has declined by more than 70 % in north-eastern Spain (Maceda-Veiga et al. 2010b), with a large decline also documented in other European countries (Helfman 2007). The building of the main reservoirs was also responsible for the decline of the sea lamprey (*Petromyzon marinus*), sturgeon and shads (*Alosa* spp.) in the Guadalquivir (Southern Spain) (Granado-Lorencio 1991), and the almost extinction of the river lamprey (*Lampetra fluviatilis*) in Tagus River (Perea et al. 2011). In contrast, some hydrological structures for agricultural purposes (channels and farm pools) may also benefit endangered native species that use them as a breeding refuge (e.g. river blenny, *Salaria fluviatilis*) (Aragon Forest Guard, *personal communication*).

Introduction of non-native fish species

The Iberian Peninsula is a hotspot for exotic introductions, considered a leading cause of extinctions (Leprieur et al. 2008; Clavero et al. 2010; Leunda 2010). Nevertheless, analysis of the contribution of exotic species to the decline of native species is often difficult because the former is concurrent with habitat degradation (Didham et al. 2007). In this regard, Hermoso et al. (2011) found that the abundance of invasive species was the best single predictor of the decline of native species, regardless of their habitat degradation status. Local extinctions of native fish have been recorded in the Mediterranean area without severe habitat degradation (Clavero et al. 2010). Introduced species can replace native species through trophic interactions, hybridization, behavioural interactions or the transfer of diseases and pathogens (Leunda 2010). In addition, exotic introductions are responsible for alterations in ecosystem function (e.g. nutrient cycling) that may also lead to native species' decline (Leunda 2010).

Trophic interactions

Predation is the most documented trophic interaction between native and introduced fish species. Although the pattern is changing (Fig. 3), many exotic fish were introduced into Iberian waters to satisfy the demands of recreational fishing, either as predators (e.g. black bass, pike-perch) or forage/bait (e.g. bleak, roach) (Elvira 1995a; Elvira and Almodóvar 2001; Leunda 2010). Regardless of interconnections between watercourses, the use of native fish as bait/forage species is associated with native translocations between Catalan watersheds (Maceda-Veiga et al. 2010a). Trophic competition may appear with the translocation of native species and the release of non-piscivorous exotic fish (Gozlan et al. 2010; Cucherousset et al. 2011). For instance, an experimental assay illustrated that mosquitofish (*Gambusia holbrooki*) achieve higher foraging values than the endangered fartet (*Aphanius iberus*) and samaruc (*Valencia hispanica*) (Caiola and De Sostoa 2005).

Hybridization

The co-habitation of similar genetic species may lead to hybridization between them. The release of brown trout (*S. trutta*) from hatchery stocks was common during the past century and continues to occur in Iberia. Iberian stocks originating in Germany and

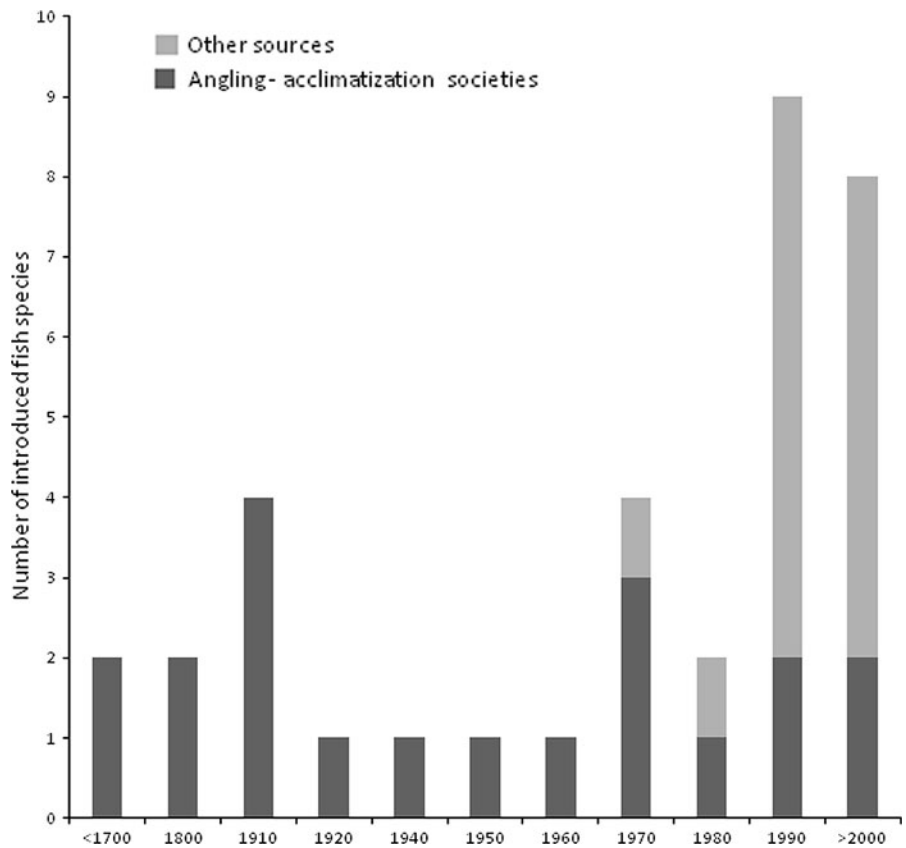

Fig. 3 Changes in the origin of exotic fish species introduced into Iberian watersheds

subsequent exchanges between hatcheries have resulted in stocks with a high degree of genetic homogeneity (Sanz and Pla 1998; Garcia-Marin et al. 1999). High levels of introgression were observed in the heavily stocked Tajo and Mediterranean basins and the lowest level was reported in the Duero basin (Almodóvar et al. 2006). Thus, the strong genetic differentiation of native trout across Iberian drainage systems was lost, due to hatchery trout stocks (Almodóvar et al. 2006). In a similar way, the translocation of native cyprinids between Iberian basins can lead to hybridization phenomena and loss of fish diversity (Doadrio 2001).

Behavioural interactions

Agonistic behaviour between native and introduced fish species has been poorly assessed (Caiola and De Sostoa 2005; Alcaraz et al. 2008; Gozlan et al. 2010). Exotics may limit the use of particular habitats or food sources by native species. An experimental assay using the native Ebro nase (*Parachondrostoma miegii*) and the exotic bleak (*Alburnus alburnus*) revealed that the bleak reduces the movement of the Ebro nase and that the former is the best trophic competitor in mixed co-habitation assays (Vinyoles et al. 2008). Another experimental assay found that native cyprinodontids increased their defensive behaviour and captured more prey than the exotic mosquitofish (*G. holbrooki*) as salinity increased (Alcaraz et al. 2008).

Transfer of diseases, pathogens and other pests

Exotic fish species may be vectors of pathogens that remain undetected until they cause clear symptomatic fish diseases or mass mortality (Gozlan et al. 2005, 2010; Maceda-Veiga et al. 2009). Although fish disease is also a cause of population decline, the study of its incidence in wild fish populations has received less attention than in commercial fish species (García-Berthou et al. 2007; Maceda-Veiga et al. 2009; Gozlan et al. 2010). For instance, the expanding eel trade in Asia has meant that the Asiatic nematode *Anguillicoloides crassus* is now widely distributed throughout East Asia in Japanese eels but also in American and European eels imported there to satisfy human demand (Gozlan et al. 2010). Examples of exotic parasite translocations already exist in Iberian watersheds (e.g. *Lernaea cyprinacea*, *Bothriocephalus aqueilognathi*, *Ichthyophthirius multifiliis*), but there are undoubtedly many more that have not been identified (Maceda-Veiga et al. 2009; García-Berthou et al. 2007; Gozlan et al. 2010). Up to 80 % (10) of the exotic parasites (12) found in Iberian fish are native to Asia (García-Berthou et al. 2007; Maceda-Veiga et al. 2009). Of particular concern is the possible introduction of *Sphaerothecum destruens* via the topmouth gudgeon (*Pseudorasbora parva*), which is already established in Iberia (García-Berthou et al. 2007; Gozlan et al. 2010). The potentially deleterious effects of this intracellular parasite on native Iberian fish species have not been assessed, but the inhibition of reproduction of several European cyprinids (e.g. *Leucaspius delineatus*) is well documented (Gozlan et al. 2005). Fish and water transfers between basins may also carry larvae or juveniles of other aquatic pests such as the zebra mussel (*Dreissena polymorpha*) or the Asiatic clam (*Corbicula fluminea*), which are already present in Iberia (García-Berthou et al. 2007).

Alterations in nutrient cycling

Alterations of native fish communities can also impair the ecosystem's nutrient cycle (Gozlan et al. 2010) (Fig. 4). The introduction of specialized plankton grazers (e.g. bleak) has a top-down effect on zooplankton that results in the absence of cladocerans, which are mainly responsible for the clear phase of lentic systems (Margalef 1983; Angeler et al. 2002). As there are no specialist grazers of Iberian native fish species, plankton have evolved without this predation pressure. The introduction of these grazers has led to poor water quality in reservoirs and can increase costs of treatment for domestic demand (Vörösmarty et al. 2010). Another example of alterations in nutrient cycling is related to the feeding activity of carp (*Cyprinus carpio*), which causes sediment resuspension with the consequent introduction of nutrients into the water column (Angeler et al. 2002; Gozlan et al. 2010; Cucherousset et al. 2011). This activity does not have a clear top-down effect on zooplankton, but resuspended sediment may damage the filter apparatus of phytoplankton grazers, and the dissolved nutrients favour algal blooms and the subsequent deterioration of water quality (Angeler et al. 2002). For instance, the presence of carp together with high temperatures may have contributed to the bloom of the cyanobacteria

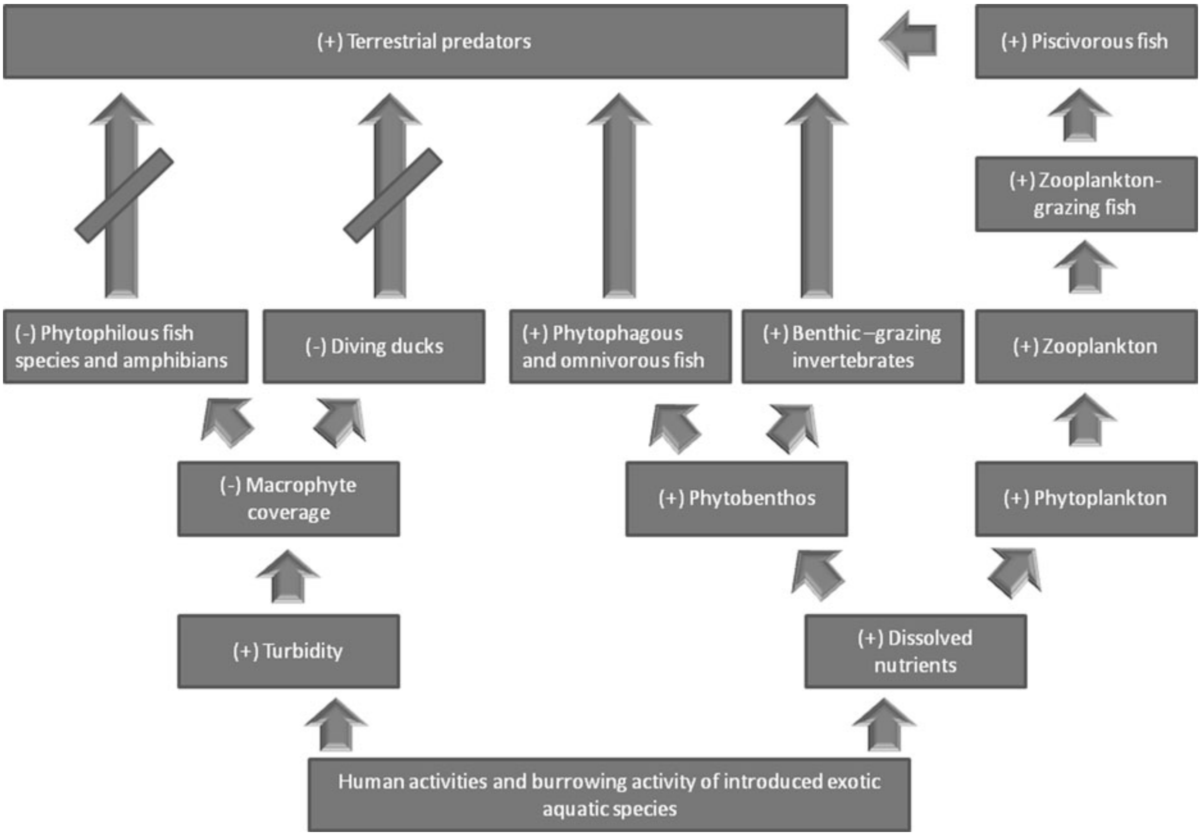

Fig. 4 Effects of human activities and the burrowing activity of introduced species on a lake in terms of food web interactions. *Arrows* indicate the energy flow within the aquatic ecosystem after increasing the dissolved nutrients and turbidity. *Crossed-out arrows* indicate the halted energy flow between compartments

Plantktothrix rubescens in Vilasouto dam (Galicia, NW of Spain), with the consequent stoppage of the water supply to nearby towns (Rodríguez, A., *personal communication*).

Water pollution

Most Iberian streams have been severely polluted by industrial, agricultural and urban activities since the mid twentieth-century (Sabater et al. 2009). The modernization of sewage treatment plants has substantially improved the chemical and biological quality of rivers in recent decades, even in low-flow Mediterranean Rivers (Prat and Munné 2000; Maceda-Veiga et al. 2010a). Nevertheless, lethal fish spill events continue to occur in Iberia, even in protected areas like the Natural Park of Sant Llorenç (Catalonia, NE Spain) (Sostoa et al. 2007) or at Alnazcóllar, close to Doñana National Park (Andalusia, Southern Spain) (Groom et al. 2006). Many of these pollutants present in sewage effluents are refractive to biological degradation (e.g. metals, pesticides, herbicides, synthetic hormones, drugs) and current sewage treatment plants fail to remove them (Lavado et al. 2004; Lavado 2006; Musolff et al. 2010; Damásio et al. 2011). Some of these compounds also reach aquatic ecosystems by atmospheric deposition and run-off from adjacent anthropic areas (i.e. diffuse pollution events) (Di Giulo and Hinton 2008; Damásio et al. 2011).

The uptake pathway and the effects that these contaminants have on fish and ecosystems (e.g. bioaccumulation, biomagnifications) depend on many factors that have been discussed elsewhere (e.g. life-history stage, trophic position of the organism, the type of xenobiotic) (Di Giulo and Hinton 2008). Examples of the effects of pollutants on fish inhabiting Iberian freshwater ecosystems can also be found elsewhere (e.g. community structure alterations, physiological dysfunctions), but few studies have focused on native fish species (Damásio et al. 2007; Raldúa et al. 2008), used biogeochemical tracers to analyse ecosystem function (i.e. effects on ecological processes) (Jardine et al. 2006) or used non-lethal sampling (e.g. blood) (Maceda-Veiga et al. 2010b). This is surprising, given the ethical questions involved and the conservation status of many native species (Doadrio 2001; Maceda-Veiga et al. 2010a). The effects of pollution on such species remain unknown when exotics are used as sentinels because ecological and biological traits differ and determine the tolerance of species to anthropogenic stressors (Di Giulo and Hinton 2008; Maceda-Veiga and De Sostoa 2011). In addition, few studies exist on the effects of personal care products and drugs on aquatic organisms, even though this is now a worldwide concern, as many of these compounds act on biochemical pathways that are common among all vertebrates (i.e. these compounds affect humans and other vertebrates in a similar way) (Musolff et al. 2010).

The input of excess nutrients from urban and agricultural sewage discharges into watercourses together with excessive pre-baiting (e.g. cereal mush release) in recreational fishing to attract cyprinids are leading causes of eutrophication (Arlinghaus et al. 2002) (Fig. 4). Many studies have examined the effects of ammonia, nitrites and a vast range of xenobiotics on aquatic species, but few have analysed the effects of nitrates on fish species, especially native Iberian fish species (Camargo 2005; Camargo and Alonso 2006). The effects of nitrates have long been underestimated, but they limit blood oxygen transport (i.e. haemoglobin oxidation) and cause endocrine disruption in fish and amphibian populations (Camargo and Alonso 2006; Edwards and Guillette 2007).

Up to 19 % of groundwater and over 50 % of water stored in reservoirs are affected by eutrophication in Iberia (Sabater et al. 2009). Together with the decrease in water transparency, algal blooms may also be responsible for oxygen depletion at night. The widespread death of fish is probably the most dramatic manifestation of hypoxia (or anoxia) and the production of highly toxic compounds (e.g. hydrogen sulphide (H_2S)) in eutrophic aquatic ecosystems (Camargo and Alonso 2006). The proliferation of cyanobacteria is also of particular concern for freshwater managers because phytotoxins (i.e. microcystins) have repercussions on both human and fish health (Margalef 1983). The potentially toxic cyanobacteria (e.g. *Anabaena*, *Aphanizomenon*, *Microcystis*, *Nodularia* and *Planktothrix*) are widespread in Spanish reservoirs and have dominated the phytoplankton community at least once in a high proportion of investigated reservoirs (35–48 %) (Quesada et al. 2004; Camargo and Alonso 2006). However, nutrient enrichment may also enhance phytobenthic biomass (Fig. 4), which may favour phytophagous and omnivorous species like the Ebro nase (*P. miegii*) and the Ebro chub (*Squalius laietanus*), respectively.

Fishery and aquaculture activities

Inland fisheries are not directly a major threat in the Mediterranean basin, unlike other regions of the world where freshwater fish are an important source of protein (Hermoso and Clavero 2011). However, excessive harvesting is contributing to the decline of some prized recreational and commercial fish species (e.g. *S. trutta*, *A. anguilla*, *P. marinus*) and almost caused the extinction of the Atlantic sturgeon (Doadrio 2001). Recreational fishing is more important (i.e. it involves millions of people) than inland commercial fisheries and contributes substantial social and economic benefits to local and national economies (e.g. fishing industries close to reservoirs) (Arlinghaus et al. 2002; Lewin et al. 2006). In the absence of official records, the volume of captures and the contribution of this recreational fishing to the economy are difficult to calculate (Cowx 2000; Arlinghaus et al. 2002). As stated in the sections above, this activity is also the main cause of fish introduction and has several other collateral effects on aquatic ecosystems (e.g. eutrophication, introduction of bait and game species) (Lewin et al. 2006).

Salmonid aquaculture is widespread in Iberia and deleterious effects on brown trout have already been documented (i.e. genetic introgression), together with the introduction of exotic salmonid species is (not a [e.g. rainbow trout, *Onchorhynchus mykiss*) due to escapes (Elvira and Almodóvar 2001; Doadrio 2001). Aquaculture research centres and ornamental aquaculture facilities have also been responsible for the introduction of exotic species (e.g. Mummichog (*Fundulus heteroclitus*), Oriental Weather loach (*Misgurnus anguillicaudatus*), Topmouth gudgeon (*P. parva*)] in the Ebro Delta (Catalonia, NE Spain). Environmental degradation caused by effluent release or the spread of diseases and pathogens (e.g. *Gyrodactylus salaris*, *S. destruens*) in native populations is also possible (Arlinghaus et al. 2002; Gozlan et al. 2010).

Alteration of riparian coverage

Human settlements have existed along watercourses in Iberia from ancient times and riparian zones have been extensively deforested and converted to agricultural, urban and industrial areas (Sabater et al. 2009). Fish thus live in a mosaic of non-forested and forested areas with plantations of native and introduced species that greatly affect river function (Naiman and Décamps 1997; Elosegui et al. 2011). The ecological importance of riparian coverage by rivers and its direct effect on fish are discussed in detail elsewhere. Riparian coverage is linked to the following: intercepting sediments, reducing erosion, acting as a green filter (i.e. removing contaminants), controlling water temperature and filtering UV radiation, providing nutrients and contributing to habitat diversification (Sagar and Glova 1995; Naiman and Décamps 1997; Elosegui et al. 2011; Flores et al. 2011).

As well as affecting river function, large exotic plantations may also influence the supply of fish food. Indeed, tree cover may be less important than prey availability in determining habitat choice by fish (Sagar and Glova 1995). Extensive eucalyptus (*Eucalyptus globulus*) and Monterrey pine (*Pinus radiata*) plantations in northwestern Iberia have changed the timing, quality (i.e. C:N:P ratio) and quantity of litter input into streams. For instance, litter input from eucalyptus plantations occurs throughout the year, with a peak in summer, although the litter is less diverse and total input may be smaller and of lower nutritional quality (lower N and P content) than in native deciduous forests (Ferreira et al. 2006). This may have repercussions on invertebrate communities and fish productivity. In this regard, deciduous forest streams contain a greater diversity of invertebrates than eucalyptus forest streams (Abelho and Graça 1996; Ferreira et al. 2006).

Ruderal plant species like the Stinging Nettle (*Urtica dioica*) or the elmleaf blackberry (*Rubus ulmifolius*) have replaced native riparian coverage in areas with high anthropic impact. The invasive Giant reed (*Arundo donax*) has also colonized extensive riparian areas of Mediterranean rivers (Munné et al. 2003). Detailed studies on the effects of Giant reed on aquatic ecosystems and fish communities are lacking. It causes physical obstructions to water flow, can cover the river bed during drought, contains lower arthropod diversity and evapotranspires three times more than native riparian species (Coffman 2007).

The periodic removal of riparian shading (e.g. logging trees, eradication of Giant reed) changes the source of energy in the stretches affected from forest inputs to primary producers present in the river (i.e. algal biomass), which may result in substantial oxygen depletion at night (Sabater 2008). Large unshaded areas may have a direct long-term effect on fish populations. Studies of salmonids reported high mortality of eggs, embryos and larvae, and the appearance of cataracts in adult fish, after continuous exposure to high but natural levels of solar UV-B (Helfman 2007). The loss of shading may also increase water temperature by up to 8 °C, which may exacerbate the effects of climate change on aquatic ecosystems (Johnson and Jones 2000). Coldwater species, like the endangered Pyrenean sculpin (*Cottus hispaniolensis*) or the brown trout, which have a thermal range below 20 °C, will be the most severely affected by such changes (Sostoa 1990; Moyle et al. 2011). These extractive riparian practices may also have other collateral effects on aquatic communities (e.g. increased run-off and siltation, use of herbicides) (Beschta et al. 2004; Helfman 2007).

Forest fires

Forest fire is a natural phenomenon that can have great ecological, economic and sociological repercussions (Otero et al. 2011; Pettit and Naiman 2007). The abandonment of forest and agricultural practices in rural areas has led to an increase in fuel and fire risk in Iberia, mainly due to involuntary and/or criminal human actions (Otero et al. 2011).

Fire may cause the following changes in aquatic ecosystems: decreased stream channel stability, greater and more variable discharge, altered coarse woody debris delivery and storage, increased nutrient availability, higher sediment delivery and transport, increased solar radiation, altered water temperature regimes and the release of toxic compounds (e.g. flame retardants, polycyclic aromatic hydrocarbons) (Rieman and Clayton 1997; Dunham et al. 2003; Vila-Escalé et al. 2007; Raldúa et al. 2008). The effects of fire on aquatic ecosystems may also depend on the intensity of the fire itself, pre- and post-fire forest management practices, natural climate constraints (i.e. floods cause soil erosion and run-off into the river) and the presence of other anthropogenic pressures (e.g. pollution events, weirs) that also constrain aquatic community resilience (Dunham et al. 2003; Sostoa et al. 2003; Pettit and Naiman 2007). There is limited understanding of the short- and long-term effects of fire on fish, which makes assessment of the risks and benefits of fire management practices pre- and post-fire difficult (Rieman and Clayton 1997; Dunham et al. 2003; Raldúa et al. 2008; Otero et al. 2011; Pettit and Naiman 2007).

The immediate consequence of a forest fire in the Natural Park of Sant Llorenç (Catalonia, NE Spain) was a reduction in fish abundance within the burnt study area (Sostoa et al. 2003). The stocks of the two native fish species (Western Mediterranean barbel, *Barbus meridionalis*; Ebro chub, *S. laietanus*) remained in the unburnt areas, but post-fire recolonization of fishless stretches had not occurred 4 years after the fire due to the existence of physical (i.e. weirs) and chemical (i.e. pollution input) barriers (Sostoa et al. 2003). The resilience of native fish to fire seems to increase when the fire affects larger interconnected systems and highly mobile species (e.g. salmonids) (Rieman and Clayton 1997). In this regard, small Iberian barbels have reduced mobility even during the spawning season (Aparicio and De Sostoa 1999). Local extirpation of native species with patchy distribution and restricted habitat requirements was also documented in North American rivers (Dunham et al. 2003). Nevertheless, the native community was re-established within a year through the return of migratory individuals that were presumably outside the system during the fire and related disturbances (Rieman and Clayton 1997; Dunham et al. 2003).

In contrast, the number of introduced species in St. Llorenç Natural Park dropped from five to one (i.e. carp, *C. carpio*), which remained in downstream stretches of the unburnt area (Sostoa et al. 2003). Similarly, post-fire floods containing high levels of suspended sediment were responsible for the elimination of the non-native brook trout (*Salvelinus fontinalis*) and rainbow trout (*O. mykiss*) in Arizona (Dunham et al. 2003). However, the effect on exotic fish species is difficult to predict because humans may aid recolonization after fire. As suggested by Moyle and Light (1996), invasion by non-native fish is inevitable if the abiotic environment is suitable for their reproduction and growth.

Gravel extraction

Mining in gravel bed streams irreversibly disturbs the hydrological and ecological function of watersheds (e.g. silting, changes in river bank morphology and pool availability), but its direct effects on fish populations are relatively unknown (e.g. Coté et al. 1999). There is much concern over this issue, as a decline in lithophilous species (e.g. nase) was documented in Switzerland after arid extraction (Zbinden and Maier 1996) and Iberian ichthyofauna include many lithophilous spawners (e.g. brown trout, chub, barbels) (Doadrio 2001; Appendix 1). Concerns are heightened when such mining affects endangered species with specific habitat requirements, such as the river blenny (*S. fluviatilis*) (Sostoa 1990). Gravel extraction may reduce the suitability of the habitat for breeding (i.e. small stones), with egg production being more strongly affected than nesting density (Coté et al. 1999). Thus, the long-term viability of these fish populations could be endangered in the absence of any positive density-dependent effects on juvenile survival. Fortunately however, *S. fluviatilis* shows a degree of habitat plasticity, which is seen in the existence of breeding populations in lakes, agricultural pools and reservoirs (Doadrio 2001; *author's personal observation*).

Management actions

European States are required to protect and restore all aquatic ecosystems with the aim of achieving good ecological status by 2015 (European Commission 2000). As common threats are affecting Iberian fish species, economic resources for conservation purposes should be allocated to improve ecological processes rather than the conservation of single species. However, the success of management actions requires good biological and ecological understanding of the relevant species. For instance, sedentary fish species like red-tailed barbels are likely to respond to local habitat restoration. To contrast, spawning migratory species like trout or large barbel species will require the restoration of residential habitats and longitudinal connectivity between adult, spawning and larval habitats, if populations are to recover (Lake et al. 2007). In any case, a domino effect (i.e. iterative effects) often exists in aquatic ecosystems and a targeted group of species (e.g. other fish, aquatic mammals, birds) and several river compartments may be benefited after the removal of main pressures (Fig. 5) (Poff and Allan 1995; Rieman and Clayton 1997; Araujo and Ramos 2000; Dunham et al. 2003). Following the IUCN criteria, the management tools considered necessary to reverse the situation of Iberian species include habitat management and restoration, the creation of protected areas and in situ and ex-situ conservation plans for particular species (Table 1).

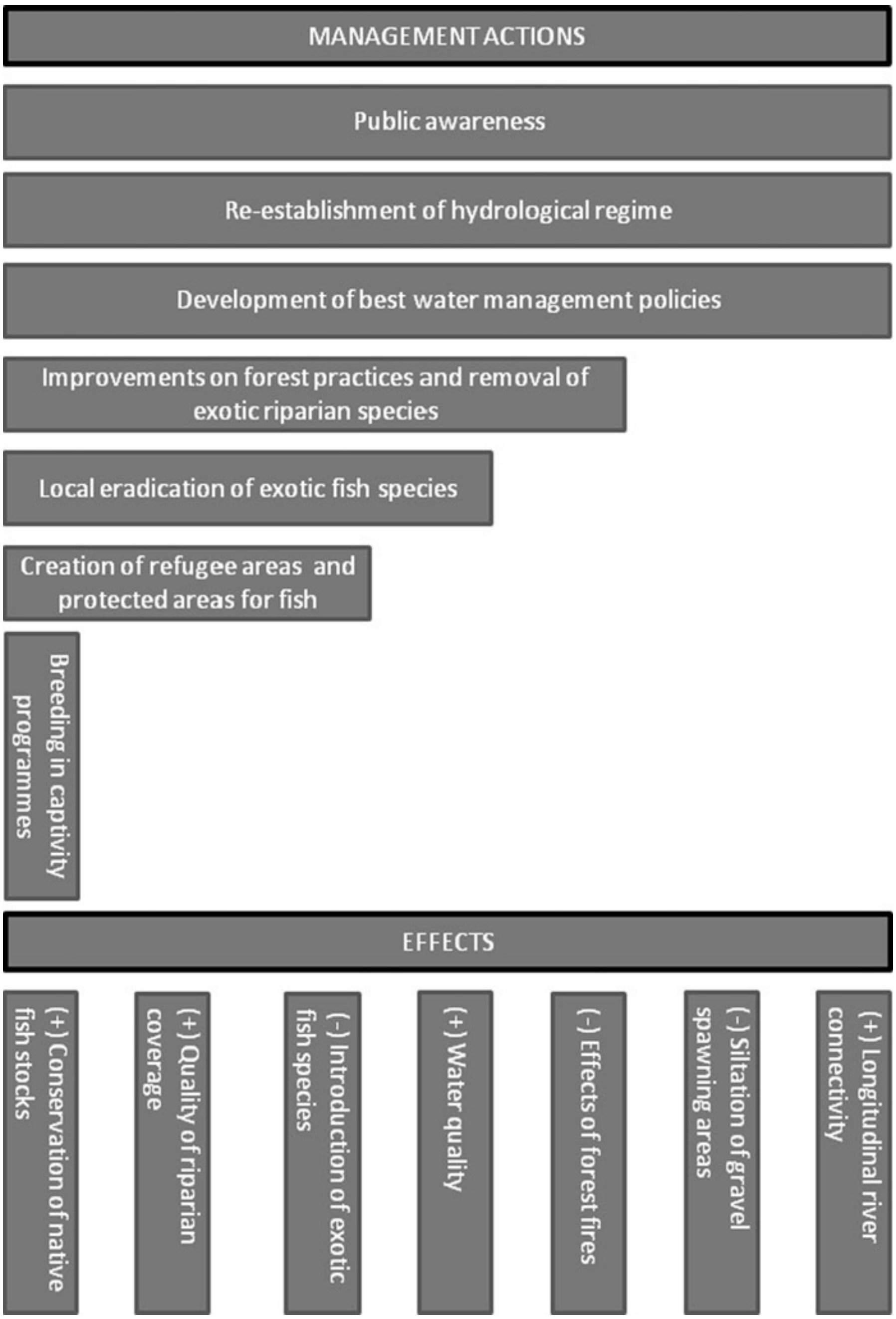

Fig. 5 Effects of several management actions on the aquatic ecosystem. The width of each *box* indicates which threats the management action will have evident positive (+) or negative (−) effects on

Habitat management and restoration

Hydrological infrastructures and water extraction

Hydrological impoundments and water extractions are the main factors responsible for river connectivity loss in Iberian rivers (i.e. habitat fragmentation). Water flow intermittency is expected to increase with climate change due to the subsequent increase in human water demands (Vörösmarty et al. 2010; Hermoso and Clavero 2011). Hydrological impoundment removal needs to be studied carefully because these physical barriers also confine exotic populations (Leprieur et al. 2006). A dam removal often has high social and political consequences and the subsequent restoration of native fish communities is not always guaranteed (Gregory et al. 2002; Johnson et al. 2008). Removal of physical barriers would be particularly useful in small tributaries with obsolete dams, in which pristine native fish populations may persist (Aparicio et al. 2000; Maceda-Veiga et al. 2010b). Fish passes can help fish overcome these physical barriers (Nicola et al. 1996; Elvira and Almodóvar 2001; Santos et al. 2002; Helfman 2007). However, for instance, only 8 % of obstacles in Catalonia (NE Spain) have fish passes and there are doubts about the usefulness of their design, as there is about those installed in other Iberian watersheds (i.e. species-specific, low water flow, poor maintenance) (Ordeix et al. 2011). These fish ladders are suitable for small physical barriers, but not for large dams (Nicola et al. 1996; Olden et al. 2011). In such cases, migratory species need to be transported via terrestrial transport (e.g. eels at the Belesar dam, Galicia, NW of Spain) (López-Ares, *personal communication*) or fish lifts (e.g. the Limia basin in Portugal) (Santos et al. 2002). Although these management actions may assist in fish conservation, the hydrological alterations remain. Experimental water and sediment releases would be the best management option for large dams to guarantee river function (Rovira and Ibañez 2007). Flow discharges should be adjusted according to the life history of the species present, rather than simply establishing some minimum flow requirement (Helfman 2007; Hermoso and Clavero 2011). This is particularly important in Mediterranean Rivers, in which strong and unpredictable hydrologic fluctuations occur naturally (Gasith and Resh 1999; Hermoso and Clavero 2011).

Best water management policies based on the reduction of water extraction (e.g. changing irrigation methods) and water re-utilization are also needed. Such measures would avoid the construction of more dams and the interconnection of waterways, with their associated sociological and ecological repercussions (e.g. regional human troubles, translocation of native species, spread of exotic species, diseases and pathogens); and to guarantee the dilution capacity of rivers and restore their longitudinal functionality (Prat and Munné 2000; Bromley et al. 2001; Dudgeon et al. 2006; Lewin et al. 2006; Johnson et al. 2008; Maceda-Veiga et al. 2010b; Vörösmarty et al. 2010; Hermoso and Clavero 2011). The re-establishment of river connectivity would have direct improvements on migratory fish species movements, with subsequent repercussions on the whole ecosystem (i.e. transfer of energy along the watercourse) and collateral benefits to other species (i.e. dispersion of glochidia of some freshwater clams) (Araujo and Ramos 2000; Helfman 2007). The absence of *A. sturio* specimens from Iberia for many years is a likely cause of the decline of the giant pearl mussel *Margarifiera auricularia* (Araujo and Ramos 2000). River connectivity is also crucial for the recolonization of fishless areas from donor populations after high-impact disturbance (e.g. forest fires, spills) (Nicola et al. 1996; Gregory et al. 2002; Helfman 2007; Olden et al. 2011).

Dealing with non-native species

Human involvement converts the introduction of exotic species into aquatic systems into a social, economic and sometimes unpredictable ecological problem (García-Berthou et al. 2007; Gozlan et al. 2010; Cucherousset et al. 2011). Further studies are needed to clarify the ecological processes involved in invasions (e.g. trophic competition by combining traditional gut content and stable isotope analyses) and to establish the incidence of diseases in wild freshwater fish populations (Elvira and Almodóvar 2001; García-Berthou 2007; Maceda-Veiga et al. 2009; Gozlan et al. 2010; Cucherousset et al. 2011). However, exotics may now have become key trophic resources in some watersheds, and their disappearance may affect the populations of other valuable species (e.g. endangered mammals, birds) (Groom et al. 2006; Tablado et al. 2010).

Complete eradication once exotics are established and widespread is a utopia and highly costly. Nevertheless, local exotic eradication plans have a more balanced cost-success ratio and are particularly useful when they are applied in small areas of high conservation value (Aparicio et al. 2000; Clavero et al. 2004; Maceda-Veiga et al. 2010b). For instance, the local extirpation of carp (*C. carpio*) with rotenone (Legumine®) in two consecutive applications with 6 days lap (90 and 50 ppb, respectively) has already been successful in Zoñar lagoon (Andalusia, Southern Spain), which is inhabited by *Atherina boyeri* (Fernández-Delgado 2007). After carp removal, water transparency increased, macrophytes were also detected and diving ducks [e.g. Common Pochard (*Aythya ferina*), White-headed duck (*Oxyura leucocephala*) and Little Grebe (*Trachybaptus ruficollis*)] returned to the lagoon. Prior to the treatment, the piscivorous Grey Heron (*Ardea cinerea*) and Great Cormorant (*Phalacrocorax carbo*) dominated the water-bird community. To eradicate exotics in current waters, it seems more prudent to use other methods such as successive electrofishing passes, nets and traps.

Re-establishment of the natural hydrological regime may also hinder the spread and establishment of exotic species (i.e. limnophilous species) in current waters (Poff and Allan 1995; Elvira and Almodóvar 2001; Clavero et al. 2004; Maceda-Veiga et al. 2010b). Moyle and Light (1996) argued that the importance of biotic resistance to invasion is low when environmental conditions favour exotics. Healthy native fish communities may resist invasion when the introduced species are non-piscivorous because the native species might use the natural resources present more efficiently (Gozlan et al. 2010). As Iberian fish species have evolved under low predation pressures, they are especially vulnerable to exotic predator fish species (Doadrio 2001; Elvira and Almodóvar 2001). Particular environmental conditions sometimes act as a natural refuge for endangered species. In Spanish wetlands, mosquitofish peak at intermediate salinities and are rare in coastal lagoons with salinities >15 ‰, where they are replaced by the Spanish toothcarp *A. iberus*, which has mainly been extirpated from fresh and oligosaline waters (Alcaraz et al. 2008). Nevertheless, the recent introduction of another hyperhaline-tolerant species, the mummichog (*F. heteroclitus*), is a new threat to *A. iberus* populations (García-Berthou et al. 2007).

Prevention is the best way to avoid further introductions into watersheds. Education programmes, together with the banned exploitation of recognized invaders, are key steps towards achieving this goal (Elvira and Almodóvar 2001; Helfman 2007; Olden et al. 2010). For instance, people believe that Eastern mosquitofish (*G. holbrooki*) is the best mosquito controller, although *Aphanius* genus show similar predation rates (Homski et al. 1994). Biological control practices need caution because they often result in the introduction of new invaders (e.g. *G. holbrooki* for mosquito control or grass carp, *Ctenopharyngodon idella*, for vegetation control) (Elvira and Almodóvar 2001; Gozlan et al. 2010). In addition, some management practices planned to benefit native species may threaten them by limiting trophic resources at the larval and juvenile stages. Pyrenean minnows (*Phoxinus bigerri* and similar species) are released into watercourses as a food supply to enhance brown trout catches. However, the juvenile brown trout (<20 cm) and the minnows, which form dense shoals, both forage mainly on the same prey items (i.e. trophic competition) (Oscoz et al. 2008).

The educational task is more difficult when exotics already enhance local or regional economies. For instance, recreational fish industries adjacent to large reservoirs or lakes are now fully developed for the exploitation of exotic fish species, including fishing competitions (e.g. black bass in Ebro reservoirs). Current inland fisheries legislation needs to be applied evenly through all Iberian regions to face common issues like the introduction of exotic species (e.g. euthanize and banned live-bait and pre-baiting activities). In this regard, fisheries regulation, involving restrictions on the number of captures and the increased length of removed fish in Iberian watersheds, have had a certain amount of success in some native species like the brown trout (Almodóvar et al. 2006; Araguas et al. 2009). It is also worthwhile encouraging anglers to practice less invasive recreational fishing (e.g. catch-release practices), although substantial post-release mortality and reduction in the growth and fitness of released specimens have also been documented (Arlinghaus et al. 2002).

Together with angling activities, attention should be paid to aquaculture facilities (research, ornamental and food) and releases of unwanted pets, because these are emerging as important pathways for freshwater introductions and have received minimal attention from regulatory agencies (Olden et al. 2010; author *unpublished data*). Perhaps everyone who handles wild and captive fish (e.g. anglers, employees of ornamental fish stores) should receive training and have to obtain a permit for handling and/or selling these species. To prevent further invasions, predictive models may also help managers to identify potential invaders before they are introduced (García-Berthou 2007). Thermal tolerance is a key parameter in these models that are region-specific (García-Berthou 2007). Special attention should be paid to regions with a wide range of climatic conditions (i.e. north-western Iberia is colder than the south); and the effects of climate change, which will favour the introduction and spread of tropical species, should be taken into account.

Water pollution

Acute and diffuse water pollution are the main reasons for habitat degradation and, as stated above, their consequences are exacerbated in arid and semi-arid ecosystems by the reduced capacity of rivers to dilute because of climatic and anthropic constraints (i.e. water extractions) (Hermoso and Clavero 2011; Grantham et al. 2012). Climate change could increase these deleterious effects on aquatic organisms because high water temperatures increase the metabolism of poikilotherm organisms (e.g. fish) (Helfman 2007; Di Giulo and Hinton 2008). Diffuse water pollution can be reduced by encouraging farmers to be more ecological or by introducing economic incentives to persuade them to produce more ecological crops. The eradication of acute water pollutant events mainly requires the removal of illegal collectors, making sewage treatment plants big enough to handle the input flow of sewage and installing in these plants green filters (i.e. tertiary treatment), through which effluents can flow before reaching the river. However, these investments could be ineffective, in part, when the natural hydrological regime is not guaranteed and the self-depuration of rivers is limited (e.g. deterioration of riparian coverage), as is the case of Mediterranean rivers (Prat and Munné 2000).

Measuring how waste water treatment plant effluents affect aquatic communities based on river flow has important implications for the way rivers are managed in Iberia and other Mediterranean climate regions of Europe (Grantham et al. 2012). These authors suggested that the effluents from sewage treatment plants should probably be maintained at a dilution rate below 5 % of the total discharge of the receiving waters if healthy aquatic invertebrate assemblages are to be preserved. The role of river pollution awareness is also relevant to the management of effluents from sewage treatment plants. In contrast to the cheapest location downstream from towns, citizens and local farmers are more conscious of pollution events in rivers when sewage inputs are placed upstream, and the subsequent social repercussions affect water management decisions (Prat, *personal communication*). The Ripoll river (Besòs basin, Catalonia, NE Spain), which was considered one of the most polluted rivers in Europe, benefited from these changes in the position of sewage inputs and partially recovered its native fish fauna in a short period of time (2003–2007) (Sostoa et al. 2007). However, spills still occur and sub-lethal effects are evident in these fish species exposed to sewage effluents (Maceda-Veiga et al. 2010b). The creation of adjacent lagoons connected to the main stream on the downstream side could be an additional useful management action to guarantee fish survival in case of spills in these high-risk areas. Fish can swim to these refugee areas when they detect the presence of xenobiotics in the main river, and recolonize it after the water flow re-establishes the water quality. The recuperation of rivers' water quality also has collateral effects. The nutrient reduction (especially phosphorus) causes phytoplankton depletion with the subsequent increase in water transparency in the lower Ebro (Ibañez et al. 2008). In addition, hydrological regulation has removed the effect of the natural shaping agents (i.e. flooding events) in Mediterranean rivers (Hermoso and Clavero 2011). The result is the spreading of macrophyte coverage and the annoying black fly (Ibañez et al. 2008). However, this scenario is also likely to have increased the effects of predation on native estuarine species, because estuarine fish evolved in turbid waters and now become more visible to the large number of exotic predators present in this basin (Helfman 2007; Maceda-Veiga et al. 2010a).

Riparian coverage and insights into forest fires

Riparian coverage is a key component of freshwater ecosystems (e.g. Naiman and Décamps 1997), but important features of riparian forest ecology (e.g. vigour of seedling growth, size- or age-distribution, abundance of dead wood) are not properly assessed in widely applied riparian quality indices (e.g. QBR) (Munné et al. 2003; Elosegui et al. 2011). The current abandonment of farming could be combated by authorities giving economic incentives for forest restoration or directly performing the periodic clearing of riparian and forest cover and the removal of exotics (i.e. Giant reed). This would reduce the evapotranspiration rate and the risk of forest fires, especially in the Mediterranean area (Coffman 2007; Otero et al. 2011). These periodic activities clearing away the riparian canopy should only affect short and staggered stretches of stream, because major changes in aquatic communities have been reported after the removal of as little as 150 m of riparian canopy (Elosegui et al. 2011). Herbicides to eradicate exotic plantations should be used with caution. For Giant reed eradication, it was reported that an assay with the herbicide glyphosate in realistic concentrations caused certain oxidative stress and potentially genotoxic effects to the blood cells of a tolerant species like the European eel in the Llobregat basin (Catalonia, NE Spain) (Puértolas et al. 2010; Maceda-Veiga and De Sostoa 2011). The removal of these exotic plantations should be followed by riparian restoration programmes. Riparian buffers with widths ranging from 5 to 30 m have been found to be 50–75 % effective at preserving the ecological functions of pristine forest streams in Britain (Wichert and Rapport 1998).

The best ecological option in regions with tradition in forest plantations (e.g. North and NW Iberia) is to keep riparian buffer strips of deciduous native vegetation mixed with these plantations to mitigate their effect on the nutrient cycle by increasing the diversity of litter input into the streams (Abelho and Graça 1996; Ferreira et al. 2006). This forest structure without large areas of pyrophilous species (e.g. *Pinus*, *Eucalyptus*, *Arundo*) may also diminish the risk of forest fires. However, it should not be forgotten that forest fires are a natural disturbance in Mediterranean ecosystems. The possibility of recovery of burnt areas is enhanced by leaving the burnt trees, whose deep and extensive roots are much more effective than grasses and shrubs at stabilizing stream banks. In addition, burnt trees and wood piles attract birds that, by dispersing plant seeds, assist the recovery of vegetation after fire (Rost et al. 2010). Dead wood is also important for the river in creating physical habitat and recycling nutrients (Beschta et al. 2004; Elosegui et al. 2011). An experimental assay in the Basque Country (Northern Spain) used this approach and reported a low turnover of organic matter and an increase in invertebrate density and brown trout biomass and age-classes (Flores et al. 2011; Elosegui et al. 2011). To avoid excessive run-off of sediment into the river, plantations of native riparian species and the installation of transverse physical barriers may also help to stabilize the substrate prior to the recovery of riparian coverage. However, the installation of complex bank stabilizers alone may be limited in its efficacy, with high costs and a short lifetime, especially in fast-flowing streams (Beschta et al. 2004).

In situ and ex-situ conservation programmes

The strong decline of freshwater fish species implies that further ex-situ conservation plans should be implemented in the near future (Elvira 1995b; Doadrio 2001; Smith and Darwall 2005; Maceda-Veiga et al. 2010a). Genetic reservoirs and captive breeding have been used for in situ conservation and to protect remnant populations of endangered species in ex-situ conservation programs, respectively (Groom et al. 2006; Araguas et al. 2009). Small endangered fish species are often candidates for ex-situ conservation programs as they are easy to breed and maintain in captivity (i.e. *A. iberus*, *V. hispanica*, *G. aculeatus, S. fluviatilis*). The expert advice of highly qualified aquarium hobbyists (e.g. Aquarium hobbyist associations) may be helpful in this respect, although at present such groups are generally ignored in fish conservation programmes (Escribano-Alacid, *personal communication*).

A common strategy for the recovery of wild populations involves the release of individuals from captive stocks or transfer from other healthy wild populations (Groom et al. 2006; Helfman 2007; Olden et al. 2011). Nevertheless, it would be more prudent to try first to increase population size through habitat improvement and restrictive regulations for exploited populations (e.g. trout) rather than to rely on reintroduction or fish transfer from other populations (Sanz and Pla 1998; Groom et al. 2006; Olden et al. 2011). Ex-situ conservation programs are controversial when reintroduction is unlikely. In addition, breeding native species without genetic guarantees often leads to the loss of diversity within populations, introgression and eventually the extinction of local populations (Sanz and Pla 1998; Araguas et al. 2009; Helfman 2007). The selection of an adequate number of breeding individuals and consideration of the genetic peculiarities of each population are crucial in ex-situ conservation breeding programs (Sanz and Pla 1998). In addition, captive-reared populations need to have sanitary guarantees and become familiar with conditions in the wild before release (i.e. immunity activation, being able to feed on wild prey, acquisition of anti-predator behaviour) (Helfman 2007; Faria et al. 2010).

The creation of "genetic refuges" and completely protected areas have aided in the conservation of native fish (Almodóvar et al. 2006). However, though protected areas are little used in freshwater ecosystem protection, it is also worth noting that the removal of some impacts requires effective action at the basin scale and not just within a small protected area (Filipe et al. 2004; Abell et al. 2007; Helfman 2007; Nel et al. 2009). Genetic refuges, in which the release of hatchery fish is banned, have been established in the eastern Pyrenees since 1997 by the Autonomous Government of Catalonia (NE Spain) as a management strategy to preserve the remaining native gene pools of brown trout and to improve fish catches (Araguas et al. 2009). The collaboration of anglers' associations has been crucial to extending these refuges to other river basins in the Pyrenees (Araguas et al. 2009). However, species recovery is sometimes poorer than expected (Araguas et al. 2009) and sometimes high (Almodóvar et al. 2006).

Concluding remarks

The conservation of natural resources, including fish populations, is a social, political, economic and ecological concern. Effective management actions require the understanding of all ecosystem stakeholders and the application of an identical legislation evenly through all Spanish and Portugal regions to face common issues (e.g. exotic species). Conservation efforts should be focused on the preservation of ecological processes, in order to achieve the goals of the Water Framework Directive and guarantee the conservation of Iberian native fish species. Native ichthyofauna have suffered a strong decline in last decades, but some small streams still maintain pristine fish populations. In addition, freshwater ecosystems are recognized as systems open to rehabilitation procedures. This means that fish and rivers themselves can do most of the restoration work when the main pressures disappear. In situ conservation measures need to be prioritized over the ex-situ ones. The best water management policies, together with the preservation of horizontal and longitudinal connectivity, are required in Iberian Rivers (in particular, in the Mediterranean area) in order to preserve the ecosystem function and maintain the ability of fish to recolonize fishless areas after natural or anthropic catastrophes. Otherwise, natural constraints and climate change may exacerbate the consequences of current anthropogenic impact. In this context, climate change may increase the effects of water pollution on aquatic ecosystems and favour the spreading of tropical fish species in Iberia. Public awareness about the threats posed by introduced species is required to prevent further invasions. Despite the interaction of many threats to fish biodiversity, freshwater managers and researchers should not let the trees prevent them from seeing the overall wood, when seeking to achieve practical solutions with the best balanced cost benefit.

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Appendix 1

See Table 2.

Table 2 List and ecological guilds of native Iberian freshwater fish based on Doadrio 2001, Kottelat and Freyhof (2007) and Leunda et al. (2009). Conservation status and population trend were extracted from the IUCN Red List of Threatened Species (www.iucnredlist.org) in December 2011

Native fish species	Status	Spawning habitat	Feeding habitat	Trophic position	Population trend
Achondrostoma arcasii	V	LITH	WC	OMNI	Declining
Achondrostoma occidentale	E	LITH	WC	OMNI	Unknown
Achondrostoma oligolepis	LC	LITH	WC	OMNI	Unknown
Achondrostoma salmantinum	E	LITH	WC	OMNI	Declining
Acipenser sturio	CE	–	B	INVE	Declining
Alosa alosa	LC	LITH	WC	PLAN	Unknown
Alosa fallax	LC	LITH	WC	PLAN	Stable
Anaecypris hispanica	E	PHYT	WC	OMNI	Declining
Anguilla anguilla	CE	–	B	PISC	Declining
Aphanius baeticus	E	PHYT	WC	INVE	Declining
Aphanius Iberus	E	PHYT	WC	INVE	Declining
Atherina boyeri	LC	PHYT	WC	INVE	Unknown
Barbatula quignardi	LC	LITH	B	INVE	Unknown
Luciobarbus bocagei	LC	LITH	B	OMNI	Unknown
Luciobarbus comizo	V	LITH	B	OMNI	Declining
Luciobarbus graellsii	LC	LITH	B	OMNI	Unknown
Barbus guiraonis	V	LITH	B	OMNI	Declining
Barbus haasi	V	LITH	B	INVE	Declining
Barbus meridionalis	NT	LITH	B	INVE	Stable
Luciobarbus microcephalus	V	LITH	B	OMNI	Declining
Luciobarbus sclateri	LC	LITH	B	OMNI	Unknown
Luciobarbus steindachneri[a]	V	LITH	B	OMNI	Declining
Cobitis calderoni	E	LITH	B	INVE	Declining
Cobitis paludica	V	LITH	B	INVE	Declining
Cobitis vettonica	E	LITH	B	INVE	Declining
Cottus aturi	LC	LITH	B	INVE	Unknown
Cottus hispaniolensis	LC	LITH	B	INVE	Unknown
Gasterosteus aculeatus[b]	LC	PHYT	WC	INVE	Unknown
Gobio lozanoi	LC	LITH	B	INVE	Unknown
Iberochondrostoma almacai	CE	LITH	B	OMNI	Declining
Iberochondrostoma lemmingii	V	LITH	B	OMNI	Declining
Iberochondrostoma lusitanicum	ND	LITH	B	OMNI	ND
Iberochondrostoma olisiponensis	ND	LITH	B	OMNI	ND
Iberochondrostoma oretanum	CE	LITH	B	OMNI	Declining
Lampetra fluviatilis	LC	LITH	B	PLAN	Unknown
Lampetra planeri	LC	LITH	B	PLAN	Unknown
Parachondrostoma arrigonis	CE	LITH	B	PHYT	Declining
Parachondrostoma miegii	LC	LITH	B	PHYT	Stable
Parachondrostoma turiense	E	LITH	B	PHYT	Declining
Petromyzon marinus	LC	–	B	PISC	Unknown
Phoxinus bigerri	LC	LITH	WC	INVE	Unknown
Phoxinus septimaniae[c]	LC	LITH	WC	INVE	Unknown
Pseudochondrostoma duriense	V	LITH	B	OMNI	Stable

Table 2 continued

Native fish species	Status	Spawning habitat	Feeding habitat	Trophic position	Population trend
Pseudochondrostoma polylepis	LC	LITH	B	OMNI	Stable
Pseudochondrostoma willkommii	V	LITH	B	OMNI	Declining
Salaria fluviatilis	LC	LITH	B	INVE	Stable
Salmo salar	LC	LITH	WC	PISC	Unknown
Salmo trutta	LC	LITH	WC	PISC	Unknown
Squalius alburnoides complex	V	LITH	B	INVE	Stable
Squalius aradensis	V	LITH	B	OMNI	Unknown
Squalius carolitertii	LC	LITH	B	OMNI	Unknown
Squalius castellanus	E	LITH	B	OMNI	Declining
Squalius laietanus	LC	LITH	B	OMNI	Unknown
Squalius malacitanus	E	LITH	B	OMNI	Declining
Squalius palaciosi complex	CE	LITH	B	OMNI	Declining
Squalius pyrenaicus	ND	LITH	B	OMNI	Declining
Squalius torgalensis	E	LITH	B	OMNI	Declining
Squalius valentinus	V	LITH	B	OMNI	Declining
Syngnathus abaster	LC	–	B	INVE	Unknown
Tinca tinca	LC	PHYT	B	OMNI	Unknown
Valencia hispanica	CE	PHYT	WC	INVE	Declining

Conservation status categories: *CE* critically endangered, *E* endangered, *V* vulnerable, *LC* least concern, *NT* not threatened and *ND* nithout enough data for categorization; spawning habitat: *LITH* lithophilous and *PHYT* phytophilous; feeding habitat: *WC* water column and *B* benthic/benthopelagic; trophic position: *PLAN* planktophagous, *INVE* invertivorous, *OMNI* omnivorous and *PISC* piscivorous

[a] This species is not valid for some authors who consider it an ecotype of *Luciobarbus comizo* with a hybrid origin. [b] The species *Gasterosteus gymnurus* is not supported by molecular data. [c] The existence of Spanish populations requires to be confirmed by genetic tools

Appendix 2

See Table 3.

Table 3 List of exotic fish species introduced into Iberian catchments and current status based on Doadrio (2001), Kottelat and Freyhof (2007), Leunda (2010)

Introduced fish species	Status
Alburnus alburnus	Invasive
Ameiurus melas	Invasive
Australoheros facetus	Invasive
Carassius auratus	Invasive
Cyprinus carpio	Invasive
Fundulus heteroclitus	Invasive
Gambusia holbrooki	Invasive
Lepomis gibbosus	Invasive
Micropterus salmoides	Invasive
Misgurnus anguillicaudatus	Invasive
Pseudorasbora parva	Invasive
Sander lucioperca	Invasive
Silurus glanis	Invasive

Table 3 continued

Introduced fish species	Status
Abramis brama	Naturalized
Blicca bjoerkna	Naturalized
Esox lucius	Naturalized
Hucho hucho	Naturalized
Oncorhynchus kisutch	Naturalized
Oncorhynchus mykiss	Naturalized
Perca fluviatilis	Naturalized
Poecilia reticulata	Naturalized
Rutilus rutilus	Naturalized
Salvelinus fontinalis	Naturalized
Scardinius erythropthalmus	Naturalized
Acipenser baerii	Uncertain
Acipenser naccarii	Uncertain
Alburnoides bipunctatus	Uncertain
Aphanius fasciatus	Uncertain
Cobitis bilineata	Uncertain
Ictalurus punctatus	Uncertain
Lates calcarifer	Uncertain
Leuciscus idus	Uncertain
Astronotus ocellatus	Unsuccessful
Barbonymus schwanenfeldii	Unsuccessful
Ctenopharyngodon idella	Unsuccessful
Piaractus brachypomus	Unsuccessful

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