GLOBAL CHALLENGES IN RECREATIONAL FISHERIES

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Chapter 4

Biological impacts of recreational fishing resulting from exploitation, stocking and introduction

Wolf-Christian Lewin, Daryl Peter McPhee and Robert Arlinghaus

Abstract

While the biological impacts of commercial fishing are well documented, those of recreational fisheries have received less attention. However, intensive and selective angling and related activities (e.g. fish stocking and introductions) can affect fish populations and aquatic ecosystems, often in conjunction with impacts external to the fishery. The risks range from those occurring to the exploited fish population (truncation of the natural age and size structure, delay of stock rebuilding through depensatory mechanisms, loss of genetic variability and adaptation, evolutionary changes) to those that occur to the aquatic ecosystem (changes in trophic cascades or nutrient cycling). In particular, genetic changes and the loss of biodiversity can be a severe threat to fish communities and ecosystems and require sophisticated management approaches. Finally, those implications for a sustainable management of recreational fisheries are discussed, which can help to reduce or avoid unwanted biological effects, social conflicts and ensure the long-term persistence of the natural resources.

Introduction

Fishing is of worldwide importance for the generation of food, income and for the satisfaction of various non-consumptive social needs (Arlinghaus *et al.* 2002). Commercial fishing and fisheries-related activities, such as the stocking of hatchery-reared fish and the introduction of non-native fish species have been the focus of concerns with respect to stock declines and vanishing aquatic biodiversity (e.g. Myers and Worm 2004; Allan *et al.* 2005; Eby *et al.* 2006). Yet, during the last decade it has been increasingly recognized that recreational angling is often the sole or dominant use of fish stocks in many freshwater habitats and coastal areas of industrialized societies (Arlinghaus *et al.* 2002; Cooke and Cowx 2006) and that its importance is also increasing in developing countries (Cowx 2002).

Recent studies point out that both commercial fishing, but also recreational angling potentially contribute to global fish decline (McPhee *et al.* 2002; Post *et al.* 2002). This chapter focuses on exploitation, stocking and introductions, such activities associated with recreational fishing that can have long-term and sometimes irreversible direct and indirect effects on fish populations and aquatic ecosystems. Stocking is the addition of native fish to a water body, whereas introduction means the introduction of exotic, that is, non-native, fish species or the transfer of native fish species between biogeographically disconnected catchments. Direct effects are those that occur directly on the exploited fish population, whereas effects occurring on the level of food webs and ecosystems are classified as indirect effects. Impacts associated with recreational fishing such as habitat modifications, bait harvesting, wildlife disturbance, nutrient input, noise and loss of fishing tackle have mainly local (but sometimes significant) importance; they are, however, not the focus of this chapter (for details, see Lewin *et al.* 2006). It should also be noted that, in many cases, the biological impacts of angling occur in addition to substantially modified habitats and increased aquatic pollution as a result of continued urban and industrial development. Hence, non-fishery impacts can have a more severe negative influence on fish populations than recreational exploitation alone (Arlinghaus *et al.* 2002). However, this does not free the recreational fishing sector from the obligation to address the sometimes contentious issue of its own impact. On the contrary, the increased awareness that the aquatic resources are not infinite, and that there is a tight linkage between ecosystems and people on a scale that transcends traditional fisheries management boundaries, necessitates an interdisciplinary approach, particularly as the fisheries management authorities usually do not have the political or financial power to implement, for example, restoration programmes on their own (Knudsen and MacDonald 2000). Furthermore, sufficient cooperation between all or the majority of stakeholders can prevent or minimize inter- and intrasectoral conflicts (Arlinghaus 2005). For example, the cooperation between fisheries stakeholders, fisheries managers, scientists and environmental conservationists may contribute to a better understanding of the challenges and solutions that ultimately benefit both recreational fishing and environmental conservation.

The objective of the present chapter is to highlight important issues based on selected examples extracted from the literature. We first present patterns of angling exploitation and the direct consequences on ecological and evolutionary timescales. Subsequently, we follow this order for stocking and introduction, and discuss the indirect impacts of exploitation and stocking, combined. The main types of the documented impact and potential risks resulting from angling and stocking/introduction are provided in Table 4.1.

Table 4.1 Summary of angling patterns and associated direct and indirect effects and potential risks resulting from exploitation and stocking.

	Pattern	Direct ecological and genetic effects in some target species	Potential risks
Angling	High exploitation rates	Decline in catch and harvest, high population fluctuations, depensation	Population decline, loss of fisheries value, loss of populations
Angling	High selectivity for size and size-related life-history traits	Truncation of age and size structure; decrease in age and size at maturation, possibly alteration of the genetic growth potential and some related reproductive traits	Demographic bottlenecks, loss of resilience, genetic variability and evolutionary potential
Angling	Selectivity for behavioural traits	Selection pressure against boldness, aggression, changes in migration time	Altered predator–prey interactions, reduced recruitment and growth
Stocking, transfer of native bait fish	Addition of native fish species	Increase of competition or predation, hybridization between hatchery and wild fish, outbreeding, transfer of parasites and diseases	Decrease or loss of native species, changes in the fish community, loss of genetic diversity and adaptive potential
Introduction, transfer of exotic bait fish	Introduction of non-indigenous species or transfer of fish across catchments	Increase of competition, predation, hybridization, transfer of parasites and diseases, outbreeding	Decrease or loss of native fish, amphibian, zooplankton and zoobenthon species
Indirect effects on food webs and ecosystems			
Angling	Selectivity for species	Selective removal of species, loss of biodiversity	Changes in food webs, trophic cascades and nutrient cycling, loss of ecosystem resilience
Stocking and introduction	Addition of piscivorous fish	Increased top-down control, alteration of food-web structure	Decrease of species richness and biomass of non-piscivores, changes in the zooplankton and phytoplankton community, reduction of phytoplankton biomass, changes in nutrient cycling within aquatic ecosystems, between aquatic and terrestrial ecosystems and in global nutrient cycling, loss of ecosystem resilience

Patterns of angling exploitation: exploitation rates and selectivity

Angling adds a further trophic level to aquatic ecosystems, and anglers can be regarded as key predators in aquatic ecosystems (Hilborn and Walters 1992). Although the level of angling exploitation varies locally, the angling mortality can be significantly high for some components of the food web, in particular for highly valued fish species (often large, aquatic top predators such as salmonids or percids receive a disproportional higher fishing mortality than others) (Post *et al.* 2002). Moreover, angling is highly selective not only for species but also for sizes, ages, sexes and morphological, physiological, behavioural and life-history associated traits (reviewed by Lewin *et al.* 2006). Because of trophy fishing, minimum-length regulations, or size-specific behaviour and vulnerability to the angling tackle, larger size classes are often positively selected in recreational fisheries (Olson and Cunningham 1989).

Even if fish are released because they are protected by harvest regulations or because anglers voluntarily practice catch and release (C&R), there may be negative impacts on fish populations, given a high amount of angling effort (Muoneke and Childress 1994; Batholomew and Bohnsack 2005; Arlinghaus *et al.* 2007). Depending on fishing tackle, angler experience, fish species or other fish-related or environmental factors, C&R can cause significant stress, affect reproductive success (Cooke *et al.* 2002; Steinhart *et al.* 2004) or lead to immediate or delayed post-release mortality that ranges from close to zero to over 90% in particular situations (Muoneke and Childress 1994; Bartholomew and Bohnsack 2005). Because of the magnitude of the number of fish frequently released by anglers, even relatively low levels of post-release mortality can lead to significant levels of fishing mortality for a particular species (McPhee *et al.* 2002; Sullivan 2003). However, C&R can also be a valuable management tool to reduce the mortality and to improve the size structure of exploited populations if used properly in combination with the right species, environmental factors (e.g. water temperature) and by sufficiently skilled anglers who minimize the detrimental impacts (Anderson and Nehring 1984).

Direct effects of high exploitation rates and selectivity

Ecological effects

Because of the combination of high exploitation rates and pronounced selectivity, angling can have various ecological effects on the exploited fish species. By ecological effects we mean impacts that affect the demography and dynamics of the exploited populations. Typically, exploited populations can cope with a considerable fishing mortality because of compensatory responses that allow populations to rebound from exploitation. These compensatory responses can involve changes in growth rates, fecundity and natural mortality. However, depensatory responses, known in the ecological literature as Allee effects (Stephens and Sutherland 1999), may counteract compensation if the population size is reduced to such a great extent that group dynamics and intraspecific interactions, such as mate choice, are impaired (Stephens and Sutherland 1999). In particular, predatory fish are sensitive to depensatory effects, and those effects may prevent the recovery of these populations (Post *et al.* 2002).

Size-selective angling can shift the size and age structure towards higher percentages of younger and smaller fishes in intensively exploited fish species (Goedde and Coble 1981; Almodovar and Nicola 2004). Because of the positive relationship between fish size and various reproductive traits, such as absolute fecundity or egg and larval size at hatch, the selective removal of large/old individuals may affect the reproductive capacity of the exploited fish population (Bobko and Berkeley 2004). In addition, larger and often older fish guarantee an allocation of the reproductive output over a period of many years (Secor 2000). This pattern plays an important role in the regulation of fish recruitment (Elliott 1994), and there is much evidence that a broader age structure enhances the population's resilience to external disturbances (Heyer *et al.* 2001; Hsieh *et al.* 2006). In addition, at least in some recreationally exploited fish species (e.g. salmonids), the fish are capable of social learning from older and more experienced individuals (Brown and Laland 2003). Experiments with hatchery-reared fish have shown that the contact to trained conspecies improved, for example, feeding behaviour (Brown *et al.* 2003) or predator avoidance (Järvi and Uglem 1993).

Anglers may also induce behavioural changes in the targeted fish, the effects of which may cascade through the food web. If animals avoid a refuge area because of disturbance or predation (Frid and Dill 2002), they may be susceptible to natural predators (Crowder *et al.* 1997) or experience greater competition in suboptimal or crowded habitats. The habitat change may also have indirect effects by locally influencing the distribution and biomass of their predators as well as of their prey (Lenihan *et al.* 2001). These 'trait-mediated effects' can be important in structuring food-web interactions (Biro *et al.* 2003), but have not been investigated in a recreational fishing setting so far.

Genetic effects

Many marine and freshwater fish species targeted by anglers show a structure of genetically and phenotypically more or less distinct subpopulations (Verspoor *et al.* 2005). This structure, defined by genetically based adaptations to the local environment, and by evolutionary history, demographic processes and the level of gene flow, ensures their resilience against short- and long-term environmental changes (Ryman *et al.* 1995) and plays a critical role in sustainable fisheries

(Hilborn *et al.* 2003). In particular, populations living in uncommon or variable habitats constitute an important part of a species' gene pool (Nielsen *et al.* 2001). Demographic bottlenecks resulting from high angling exploitation may reduce the genetic variation within a population and its capacity for the retention of rare alleles by genetic drift and inbreeding (Hedrick and Miller 1992), which is particularly relevant in small, isolated populations characteristic of small freshwater ecosystems (Stockwell *et al.* 2003). The genetic variability may be lowered additionally by the removal of the largest individuals (Borrell *et al.* 2004).

Because many commercially exploited fish stocks (e.g. cod, *Gadus morhua*) have declined and failed to recover even after a significant reduction of the exploitation (Walsh *et al.* 2006), there is growing concern that selective exploitation over decades may not only result in demographic consequences for targeted and non-targeted fish species but also in evolutionary changes in some traits and life-history characters (Hutchings 2005). In fact, recent studies have pointed out that intensive and selective fishing has the potential to influence life-history parameters, behavioural traits and reproductive success (Conover and Munch 2002; Walsh *et al.* 2006). Most studies on evolutionary changes as a consequence of high and selective fishing have been concerned with commercial fishing. However, there are some indications that angling also may select for or against certain traits, provided the fishing mortality is high and the survivors represent genotypes that are less vulnerable to angling mortality and proliferate in subsequent generations. For example, anglers selectively exploited the genetically distinct early running adults and therefore altered the adaptive architecture of an Iberian Atlantic salmon (*Salmo salar*) population (Consuegra *et al.* 2005). Intensive angling may have also decreased the age at first maturation in pike (*Esox lucius*) (Diana 1983) and bluegill (*Lepomis macrochirus*) populations (Drake *et al.* 1997), and may have also contributed to a change in the age at first maturation and to a degradation of the genetic growth potential of brook trout (*Salvelinus fontinalis*) populations (Nuhfer and Alexander 1994; Magnan *et al.* 2005). Recreational fishing may continue in a region long after commercial fishing operations have ceased because fish stocks have been significantly reduced and commercial fishing is not economically viable. This may lead to persistent selection pressures acting on various traits of recreationally exploited fish species.

Patterns of fish stocking and introduction

A major impact of recreational fishing on aquatic ecosystems originates from the stocking of fishes predominantly raised in hatcheries and from deliberate or accidental introductions of non-native fish species or native fishes carrying genes that evolved in other catchments than the recipient ecosystem. Such activities have been performed on a worldwide scale often without considering the effects on the native species (Cambray 2003), and many of the large lakes in the world are threatened by the introduction and spread of non-native species (Hall and Mills 2000). The introduction of cosmopolitan non-native species have contributed to a homogenization of the world's fish fauna (Rahel 2002) and have led to severe economic losses (Pimentel *et al.* 2000).

The motives for stocking and introductions are numerous, ranging from the reintroduction of locally extinct species, the enhancement of wild fish stocks to maintain or increase fish production and the improvement of water quality, to the establishment of new recreational fisheries (Cowx 1994). Stocking can mitigate the effects of recruitment failures caused by anthropogenic activities and enhance threatened fish populations (Welcomme and Bartley 1998) and can sometimes be considered a sustainable practice (e.g. in put-and-take fisheries; Arlinghaus *et al.* 2002). It may also be considered more politically preferable to harsher regulations that further limit angling catches to support growing recreational fisheries (Travis *et al.* 1998). While stocking has initially been performed in freshwater environments, there is increasing interest in marine and estuarine stocking (Taylor *et al.* 2005).

In areas where non-native species have been stocked in the past, the effects may be irreversible if the species has become established and self-sustaining, regardless of current policies that prevent further stocking. Depending on the origin and ecology of the stocked/introduced species and the biological and environmental conditions, stocked fish, particularly those not native to the recipient system, can threaten wild fish populations, shift the natural species assemblages and change aquatic ecosystems and their links to the surrounding landscape (Eby *et al.* 2006). This also applies to species that are native to a system, if they originate from hatcheries and constitute selected genetic lines that are not adapted to the system in which they are stocked.

Because the ecological risks associated with uncontrolled stocking/introduction have been increasingly recognized, legislation in many countries has reduced stocking and restricts or prohibits the introduction of non-native species. Despite this, the stocking of hatchery-reared fish and the translocations of fishes between watersheds to enhance commercial or recreational fisheries is still common (Holmlund and Hammer 2004). There are also examples of accidental introductions associated with the stocking of an unknown mixture of species (Bischoff *et al.* 1998) or the use of live bait fish (Holthe *et al.* 2005).

Direct impacts of stocking

Ecological effects

There are many indications that competitive interactions between stocked and wild fish can affect wild fish populations (Weber and Fausch 2003). The competition risk is particularly high if both species did not evolve in sympatry, so that differences in their resource use that would ameliorate competition did not evolve (Dunham *et al.* 2002). Because of anglers' preferences for predatory fishes, stocking practices often increase the densities of predatory fish. Those shifts in the natural species assemblage can contribute to the decrease or loss of local wild fish populations, may destabilize the natural predator–prey system (Stewart *et al.* 1981) and can contribute to a simplification of food webs (Schwartz *et al.* 2006). For example, introduced predatory fish contributed to the extinction of the Australian rainbow fish (*Melanotaenia eachamensis*) in the wild (Barlow *et al.* 1987), affected native galaxids in New Zealand (Townsend and Crowl 1991), native salmonids in North America (Krueger and May 1991) as well as the endemic species *Valencia hispanica* in Spain (Planelles and Reyna 1996).

The predation by stocked fish may not only affect the other fish species but also amphibian species (Collins and Storfer 2003) and aquatic invertebrates (Knapp *et al.* 2005). Negative effects of competition and predation are likely to occur if stocking biases the natural community structure or increases the fish densities beyond the threshold set by the carrying capacity of the water body.

The introduction of benthivorous fish such as carp (*Cyprinus carpio*) can cause significant changes in aquatic ecosystems if their densities exceed a water-body-specific threshold. Their feeding activity and excretion can increase turbidity, facilitate eutrophication and contribute to a regime shift in shallow lakes (Zambrano *et al.* 2001).

An additional risk associated with stocking is the worldwide spread of diseases and parasites (Lafferty *et al.* 2004). Examples are the spread of the parasites *Gyrodactylus salaris* that infects Atlantic salmon and *Anguillicola crassus* parasiting on eel (*Anguilla anguilla*). Both were spread outside their native range mainly by an anthropogenic movement of infected fish (Johnsen and Jensen 1991; Wickström *et al.* 1998). The risk of a pathogen transfer is particularly high if the introduced species is a healthy host and the local fish community has not been in former contact with the pathogen.

Genetic effects

Stocking may lead to hybridization between fishes of different origin and to gene flow from the stocked to the local fish stock. Whereas a small gene flow between populations may increase their capability for adaptation, a larger gene flow may result in the disruption of locally adapted gene complexes followed by a fitness loss in the locally adapted population (Krueger and May 1991). It is not only the introduction of non-native fish but also the translocations of fish within their native range across disconnected watersheds that can destroy the phylogeographic structure of fish species and affect the fitness of local populations (Gilk *et al.* 2004).

Hatchery-reared fish may perform poorly under natural conditions, primarily as a consequence of hatchery shortfalls such as domestication, genetic drift or inbreeding (Brown and Day 2002). The hybridization between wild and hatchery-reared fish can affect the fitness of the wild fish population. The genetic effects are especially deleterious the more the hatchery and wild fish genetically differ. Differences in terms of genetic diversity, behaviour, morphology or physiology, resulting from selection differences between hatchery and natural environments, have been demonstrated by many researchers (e.g. Einum and Fleming 2001).

Indirect effects of exploitation and stocking on ecosystems

According to the trophic cascade theory, changes in the abundance of top predators control a cascade of trophic interactions that regulate zooplankton, algal dynamics and nutrient cycles in aquatic ecosystems (Reid *et al.* 2000). Both the selective removal or the selective stocking of specific species can have severe but contrasting impacts on aquatic communities, trophic cascades and entire ecosystems (Eby *et al.* 2006).

The consequences of species-selective angling are rarely studied, and until now, there were only few indications that angling may have the potential to induce cascading effects (Pinnegar *et al.* 2000). Nevertheless, it is likely that the over-exploitation of top predators by anglers, or alternatively the increase of top predatory fishes through stocking programmes, impacts other trophic levels. The stocking of top predators can increase the strength of the top-down control leading to shifts of the species abundance and composition in the invertebrate communities (Biggs *et al.* 2000). For example, the stocking of planktivorous salmonids to enhance the recreational fishery favoured the small-bodied zooplankton species (Donald *et al.* 2001). The reduction of the large zooplankton species can release the phytoplankton from grazing pressure and promote algal biomass. In contrast, the stocking of piscivorous fish species can reduce the abundance of planktivorous fishes, increase the abundance of large-bodied zooplankton and decrease phytoplankton biomass – an effect utilized in lake restoration techniques (e.g. Carpenter and Lathrop 1999).

Predatory fish species can influence nutrient dynamics and stoichiometry via several mechanisms: by determining the magnitude of zooplankton herbivory, the nutrient recycling rate of the zooplankton and by the nutrient recycling of fish (Elser *et al.* 2000). For example, the stocking of predators can reduce the nutrient excretion by planktivorous fish (Findlay *et al.* 2005). Predators can influence the nutrient transfer between littoral and pelagic areas if they force planktivorous fish to hide and to feed in the littoral (Okun *et al.* 2005). Also, the nutrient exchange between aquatic and adjacent terrestrial ecosystems may be affected if stocked fish increase the predation pressure on amphibians or insect larvae – an effect that may cascade through terrestrial food webs (Eby *et al.* 2006). Those effects cascading down to the level of primary producers may even change the carbon exchange between aquatic ecosystems and atmosphere (Schindler *et al.* 1997).

To sum up, given a high intensity of stocking, it is not unlikely that poorly planned stocking practices can compromise biodiversity. Although the mechanisms are still a matter of debate, there are many indications that biodiversity contributes to the functioning and the resilience of ecosystems (Hooper *et al.* 2005).

Irrespective of the preceding discussion, stocking and angling exploitation do not necessarily impact fish populations negatively. There are also studies that have failed to detect any significant impact of, for example, angling mortality, if mortality levels were low or compensatory processes strong. Concerning stocking, several studies indicate that stocking, depending on environmental conditions, stocking practices and the size and origin of the stocked fish may be a useful management tool for re-establishing fish stocks (Schram *et al.* 1999) and for improving stocks of salmonids, for example (Aprahamian *et al.* 2003), eel (Leopold and Bninska 1984), tench, *Tinca tinca* (Skrzypczak and Mamcarz 2006), muskellunge, *Esox masquinongy* (Wahl 1999) and pikeperch, *Stizostedium vitreum* (Pratt and Fox 2003), as long as the fundamental problems that have caused the population decline have been addressed. In addition, stocking of piscivorous fish combined with a reduction of nutrient loads can be used to rehabilitate eutrophied lakes (Mehner *et al.* 2004). Stocking offers conflicting potentials for fisheries management and the significant ecological risks associated with stocking; in particular, the genetic impacts on wild populations need to be addressed by an appropriate risk assessment approach (Holmlund and Hammer 2004).

Implications for the management of recreational fisheries

Although the main threats to fish populations (e.g. modifications of water bodies and habitat loss) most often originate from outside the fishery (Arlinghaus *et al.* 2002), there is growing evidence that intensive angling and associated activities can have a negative impact on fish population ecosystems, contributing to regime shifts from a desired to a less desired state and thus resulting in high costs to society. Common management approaches – although often successful – cannot always prevent unwanted biological effects. The open-access nature of many recreational fisheries presents a considerable challenge to fisheries managers (Kearney 2001). To avoid or reduce biological impacts on fish stocks and ecosystems, management principles such as precaution, cooperation across stakeholders and public agencies, risk assessment, active adaptive management practices and post-implementation evaluation of management measures should be adapted and routinely applied to the management of recreational fishing (Arlinghaus *et al.* 2002).

To avoid the detrimental impacts of selective exploitation, the exploitation should not be biased towards particular components of a population. The protection of a naturally variable size and age structure and an increase of the proportion of old repeat spawners reduce the risk of recruitment failures (Birkeland and Dayton 2005). This may necessitate a greater use of maximum-size limits or slot limits (both a minimum- and maximum-size limit) for some species. Where necessary, a reduction of the overall fishing mortality may be locally needed (Post *et al.* 2002) by closing fishing or implementing successful C&R protocols. A combination of output (catch and harvest) and input (fishing effort) controls might ensure successful fisheries management. However, the effectiveness of common regulations varies to such an extent that the reliance on bag limits or minimum-size limits needs rethinking (Conover and Munch 2002).

Catch and release may help to reduce the overall fishing mortality and to preserve the natural age and size structure of exploited fish species if used properly with regard to species and environmental factors, and by sufficiently skilled anglers (Cooke and Suski 2005). Under conditions of unsustainable exploitation, other possible management actions are specific quota, access restrictions, increases in access costs in terms of time or money, lottery systems of access, annual rotation access schemes, regulations of total angling effort, total harvest limitations or combinations of these options (Carpenter and Brock 2004). The establishment of protected areas becomes increasingly important in the management of marine fisheries. With regard to inland freshwater reserves, there are very few scientific studies available, although work is emerging in the Caspian Sea region, highlighting the need to consider marine and freshwater areas in integrated reserve network planning (Khanmohammadi *et al.* in press). The protection of certain areas of rivers or lakes may contribute to the rebuilding of fish stocks, especially for fish species that are affected by fishing, habitat loss and other threats to aquatic biodiversity (Suski and Cooke 2006). To ensure the conservation of ecosystems and fish communities in the long term, protected areas must be complemented by parallel policies taken in areas outside the reserves.

Because fish populations of different water bodies vary locally and anglers link spatially segregated fisheries in space and time (Post *et al.* 2002), recreational fisheries management should be flexible, and temporally and spatially matched to the scales of ongoing exploitation, maintaining an intermediate level of aggregation of management between the extremes of one-size-fits-all and completely disaggregated management (Carpenter and Brock 2004).

The protection of genetic variability and the prevention of detrimental genetic changes are crucial for sustainable recreational fishery. Genetic aspects are particularly important in the case of stocking practices for which numerous guidelines are available (e.g. Molony *et al.* 2003; Mehner *et al.* 2004). Unjustified transfers of fish species between disconnected watersheds should be avoided. The use of local brood stocks can avoid a disruption of local adaptation. A hatchery management regime with appropriate mating schemes, breeding conditions and more natural conditions can foster the local retention of adaptive genetic variation, reduce domestication and increase the post-release survival (Brown and

Day 2002). In accordance with the precautionary approach, stocking should be accompanied by risk assessments and appropriate monitoring programmes.

However, stocking has often been considered unsuccessful for various reasons, ecological as well as economic (Moyle *et al.* 1986). In addition, stocking can make it difficult for fisheries management to monitor the status of the natural fish population and can undermine incentives for sustainable management practices (White *et al.* 1995). Consequently, a shift from a predominance of artificial stocking to the rehabilitation of aquatic habitats may be promising. The rehabilitation of aquatic habitats that aims at the ecological integrity of entire aquatic ecosystems and encompasses the increase of habitat diversity and the improvement of water quality (Roni *et al.* 2005) may be more successful than the traditional stocking practices, with ecological risks being minimal. Habitat improvement may significantly improve not only the fishery *per* se but have wider social and economic benefits to the community as a whole. Unfortunately, large-scale habitat rehabilitation projects cannot be conducted by fishery stakeholders alone, thus limiting its immediate application in recreational fisheries management. It is our hope that future efforts will strive to balance the interest of recreational fisheries with efforts to conserve fish populations in a state that could be termed 'natural'. In this respect, it is necessary to emphasize that recreational fishing is often the last factor in a chain of anthropogenic impacts on ecosystems. Therefore, effective conservation has to address all contentious issues and should focus on the most important issues in the first place, those being habitat modifications in all freshwater systems and in many coastal ones as well.

Finally, this chapter is not meant to be a blanket criticism of recreational fish exploitation and stocking. Rather, it is meant to be a summary of issues that vary in importance from locality to locality and that need to be addressed by the appropriate fisheries management. On the whole, we consider that recreational anglers can be drivers of conservation in many areas. They should not be seen as a non-natural disturbance to 'natural' ecosystems but as a component of nature that has a vested interest in preserving this situation now and in the future.

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