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Documented and Potential Biological Impacts of Recreational Fishing: Insights for Management and Conservation

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While the impacts of high exploitation on fish populations and aquatic ecosystems are well-documented for commercial fishing, particularly in the marine environment, the potential biological impacts of angling received less attention.

This paper discusses angling patterns within a framework of basic ecological and evolutionary literature and examines potential biological impacts of angling by focusing on study results associated with high exploitation rates and pronounced selective exploitation. The impacts range from impacts occurring directly on the exploited species (truncation of the natural age and size structure, depensatory mechanisms, loss of genetic variability, evolutionary changes), to those that occur on the aquatic ecosystem (changes in trophic cascades, trait-mediated effects). As a third category, impacts related to the angling activity per se are distinguished (habitat modifications, wildlife disturbance, nutrient inputs, loss of fishing gear).

Although the main threats to fish often are localized outside recreational fisheries, there is growing evidence that angling and angling associated activities can lead to a decline of fish populations and affect aquatic ecosystems in various ways provided that the degree of the fishing mortality is high and the selective exploitation is intensive. In conclusion, management implications for sustainable recreational fisheries and areas for future research are outlined.

Keywords angling, biodiversity, catch-and-release, ecological impacts, exploitation, evolutionary changes, fish community, genetic diversity, littoral habitat, recreational fishing, wildlife

Introduction

Fishing has been a popular human activity since ancient times (Yellen et al., 1995; Pringle, 1997). Up to the present day, fish populations have a worldwide importance for generating

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food, income and to satisfy various nonconsumptive social needs (Arlinghaus et al., 2002a). In addition, fish populations are part of the biocomplexity within and between ecosystems. For example, they play an important role in the functioning of ecosystems and in maintaining the diversity of regulating, linking, cultural and information services natural fish stocks provide (Holmlund and Hammer, 1999). Globally, marine and freshwater fish stocks are facing a number of threats from anthropogenic stress (e.g., exotic species invasions, hydropower generation, land use, pollution, eutrophication, habitat loss or change, river fragmentation, flow regulation, and navigation; Bruton, 1995; Cowx, 2002a; Cambray, 2003; Wolter and Arlinghaus, 2003). Many of these stresses do not originate from the fisheries sector per se (Arlinghaus et al., 2002a). Nonetheless, fishing and fisheries-related activities such as stocking of hatchery-reared fish or fish introductions can also seriously affect fish, other wildlife and ultimately entire aquatic ecosystems (Pauly et al., 2001; Robinson and Frid, 2003; Allan et al., 2005; Dobiesz et al., 2005; Frank et al., 2005; Mullon et al., 2005). Especially the loss of fish biodiversity is a matter of global concern, not only because of the associated loss of substantial intrinsic, ethical or cultural values, but also because it has numerous consequences for the human life-support systems (Holmlund and Hammer, 1999).

Most studies concerning fishing-related impacts on the aquatic environment deal with commercial fisheries. Intensive commercial fishing can lead to stock declines and species extinction (Myers and Worm, 2004; Allan et al., 2005) and induce ecosystem regime shifts (Jackson et al., 2001). However, at least the inland commercial fisheries have lost social and economic importance in the industrialized countries during the last decades. In contrast, noncommercial fishing, in particular angling, has turned out to be comparably more important, involving millions of people, thus contributing substantial social and economic benefits to local and national economis (Arlinghaus et al., 2002a; Arlinghaus and Cooke, 2005). Today, angling is the sole or dominant use of fish stocks in most freshwater habitats and in many coastal areas in the industrialized societies of the temperate regions (Welcomme, 2001; Arlinghaus et al., 2002a; Coleman et al., 2004; Cooke and Cowx, 2004, 2006). Even in the developing countries, where most fishing still focuses on food security and maximizing harvest of fish stocks for commercial or subsistence use, the importance of angling is increasing (Cowx, 2002a), e.g., for tourism development.

Athough the management of recreational fishing has regionally received considerable attention, the potential impacts of angling on fish populations and aquatic ecosystems have not been scientifically discussed to the same extent as for commercial fishing (Cooke and Cowx, 2004, 2006). The underestimation of the potential impacts of angling in fisheries research and public discussion and the focus on commercial fisheries can be attributed to several reasons. First, the dimension of angling is often underestimated. A single angler has a much lower impact on fish stocks compared to a commercial fisher operating, say, a large trawler. This perspective, however, overlooks the cumulative impacts millions of anglers can induce. The lack of representative socio-economic evaluations on the social importance of angling and the difficulty of monitoring highly diverse and disperse angling activities may contribute to the underestimation of the angling impacts (Arlinghaus et al., 2002a). Second, in contrast to commercial fishing, angling is predominantly practiced for a multitude of nonconsumptive objectives during leisure time, where catching fish for consumption is only one of many drivers for the activity (Fedler and Ditton, 1994; Policansky, 2002; Cowx, 2002b, Arlinghaus and Mehner, 2004). Therefore, angling is supposed to not be driven by the same economic incentives that drive the commercial fishery to overexploit the fish stocks (see Myers et al., 1997). This perspective, however, does not acknowledge that catching fish is the most important aspect determining the main "product" of the angling experience, satisfaction; hence catching fish matters for many anglers to a great extent (e.g., Graefe and Fedler, 1986; Petering et al., 1995; Connelly and Brown, 2000; Arlinghaus, 2006). Third, in many coastal or nearshore areas, fishery professionals and managers often focus on commercial fishing overlooking that sometimes angling can become the dominant user of fish stocks in terms of total landings (Coleman et al., 2004), and anglers also target areas not accessible to commercial fishing (Cooke and Cowx, 2004). Finally, angling is not only believed to be less harmful than commercial fishing (McPhee et al., 2002), but is often also regarded as self sustaining. Angler effort is assumed to dissipate if fish populations and subsequently the angling quality decline (e.g., Johnson and Carpenter, 1994; Hansen et al., 2000), which is not necessarily true (Post et al., 2002). Additional factors such as the lack of long-term monitoring programs, the spatial and temporal variability of fish populations, the diffuse and small structure of the recreational fisheries, the complexity of angler behavior, poor intergenerational memory of angler communities, and management actions masking the decline of fish stocks (e.g., stocking) contribute to the difficulty to obtain an accurate picture of the impacts of angling and to accept anglers as contributing, or having the potential to contribute to global fish declines (Pauly, 1995; Post et al., 2002; Cooke and Cowx, 2004; Arlinghaus and Cooke, 2005).

Recent studies have challenged the perspective that angling is more benign than commercial fishing. Available research papers focused on the impacts of angling in particular jurisdictions such as Canada (Post et al., 2002), Australia (McPhee et al., 2002) and USA (Coleman et al., 2004). These studies have highlighted that angling, if not properly managed, can negatively impact exploited fish populations. Cooke and Cowx (2004, 2006) contrasted characteristics of commercial and recreational fishing suggesting that both types of fishing share more in common than previously thought. The present review extends the previous analyses on the biological impact of angling by developing the first global review of the documented biological impacts of angling. This article focuses solely on angling and largely omits comparative statements contrasting commercial and recreational fishing. To provide a global synthesis on documented and potential impacts of recreational fishing on fish populations and aquatic ecosystems, studies on angling are supplemented by basic ecological literature and discussed within a joint framework. Thus, this review also strives to provide a new perspective on the social-ecological system "recreational fisheries" which builds on two lines of research: basic ecological research and applied research into the management and conservation of recreational fisheries. Finally, we identify knowledge gaps concerning the impacts of angling and determine management implications for sustainable recreational fisheries.

We focus on those biological impacts of recreational fishing that potentially have long term effects on fish populations and aquatic ecosystems and that might induce irreversible changes. The impacts are classified into direct and indirect impacts on exploited fish stocks as consequences of high and selective exploitation and harvest. With direct impacts we mean impacts occurring directly on the exploited target species (truncation of the natural age and size structure, delay of stock rebuilding through depensatory mechanisms, loss of genetic variability, evolutionary changes). Indirect impacts, on the other hand, involve impacts that occur via the exploitation of a target species on other components of the aquatic food webs (changes in trophic cascades, trait mediated effects). As a third category, we distinguish impacts associated with the angling activity per se, i.e., impacts not associated with exploitation and harvest of fish (e.g., habitat modifications, wildlife disturbance, nutrient inputs, loss of fishing gear).

Before the three levels of impacts are reviewed, a brief section on angling exploitation patterns follows. This is done to document major angling patterns that facilitate the biological impacts outlined later. However, it is important to note that a comprehensive review of angling patterns and the factors facilitating the effectiveness of angling (e.g., catch and exploitation rates, catchability, social dynamics) is beyond the scope of the present article.

Patterns of Angling Exploitation

Exploitation Rates

The exploitation rates (i.e., the fraction of the fish in a population at a given time that is caught and removed during a particular time interval, e.g., a year) by angling are highly variable. According to several literature reports they range from <10% to > 80% and thus can be substantial. For example, Allen et al. (1998) reported annual exploitation rates from 40–80% in black crappies (*Pomoxis nigromaculatus*), from 8–62% in white crappies (*P. annularis*) (compare also Colvin, 1991) and from 9–72% in largemouth bass (*Micropterus salmoides*). Annual exploitation rates ranged from 4–64% in bluegills (*Lepomis macrochirus*) (Beard and Kampa, 1999), from 33–48% in rainbow trout (*Oncorhynchus mykiss*) (max. 72–81%) (Cox and Walters, 2002), from 33–48% in sockeye salmon (*O. nerka*) (Matsuishi et al., 2002), from 11–54% in white bass (*Morone chrysops*) (Muoneke, 1994; Schultz and Robinson, 2002), from 7–61% in yellow perch (*Perca flavescens*) (Iserman et al., 2005) and from 4–22% in pike (*Esox lucius*) (46% for pike > 500 mm) (Pierce et al., 1995). Annual exploitation rates in walleyes (*Sander vitreus*) ranged from 7–32% (Radomski, 2003) and Mosindy et al. (1987) pointed out that anglers were able to remove 43% of the production of the adult walleye population per year.

Obviously the exploitation rate varies with angling effort (Moring, 1993; Cox and Walters, 2002; Post et al., 2003), but it is also influenced by regulations (Miranda and Allen, 2000; Beard et al., 2003a; Näslund et al., 2005), e.g., opening time and duration of the fishing period (Gunn and Sein, 2000, Margenau et al., 2003). Furthermore, it is influenced by angling gear (Smith, 2002; DuBois and Dubielzig, 2004), angler party size (Miranda, 2005), angler type (Arlinghaus, 2006), accessibility (Gunn and Sein, 2000), and travel distance (Post et al., 2002). In addition, demographic characteristics (Arlinghaus and Mehner, 2004) and other human dimensions of anglers, compliance with regulations (Sullivan, 2002), as well as experience, and specialization, perceptions, expectations, and satisfaction may have a substantial effect on exploitation (Spencer and Spangler, 1992; Wilde et al., 1998; Hunt et al., 2002; Arlinghaus and Mehner, 2004). Also environmental influences (Smith et al., 2000), season (Isermann et al., 2005), and weather conditions (e.g. Aultman and Haynes, 1993), the type and location of the water body (Miranda, 1999; Iserman et al., 2005), and the target species' ecology (Mezzera and Largiadèr, 2001; Cox and Walters, 2002) influence the exploitation rates in recreational fishing.

The interaction between anglers and an individual fish population leads to a relationship between population density of the fish stock, angler catch rate, and catchability. The catchability is the proportion of a fish population that is caught per unit angling effort (Post et al., 2002). Some studies indicate that angler catch rates are linearly related to population density and in some species such as walleye catchability was found to be density-independent

(Beard et al., 1997; Hansen et al., 2000, but see Hansen et al., 2005). The assumption that the catchability is constant or even declines with declining fish densities inter alia supports the popular view that recreational fisheries are self-regulating. However, this assumption has rarely been quantified (Pereira and Hansen, 2003) and is not neccessarily the rule in many recreationally exploited fish populations. In aggregating fish species, the catchability can increase with declining fish densities if anglers show a nonrandom searching behavior and are able to successfully locate and exploit the (shrinking) aggregations—this success being attributable to technological advances and effective communication (Peterman and Steer, 1981; Shuter et al., 1998; Smith, 1999; Post et al., 2002; Hansen et al., 2005). Behavioral (shoaling) or habitat-mediated (concentrations of fishes in spatially limited habitat) aggregations are common in freshwater fish species targeted by anglers, and the increasing catchability at declining population densities may increase the likelihood of overexploitation and contribute to an "invisible collapse" of some fish species (Post et al., 2002). Some studies found an inverse density-dependent catchability in walleye (*Sander vitreum*), Chinook salmon (*Oncorhynchus tshwytscha*), and lake trout (*Salvelinus namaycush*) fisheries (Peterman and Steer, 1981; Shuter et al., 1998; VanDeValk et al., 2005). For these fish stocks, the level of sustainable fishing is probably lower than often recognized (Cox and Walters, 2002).

Another important aspect is the feedback of anglers and fish. Anglers are often expected to abandon fishing opportunities that do not satisfy their expectation of quality angling and instead choose other recreational opportunities or shift their effort to populations with higher catch rates (Carpenter and Brock, 2004). The angler response would allow a stabilization of the exploited fish stock. However, although this expectation may often be true, the angler response depends to a high degree on the spatial scale and the elasticity of demand for fishing, e.g., angling demand on a particular water is inter alia dependent on the availability of alternative fisheries (Johnson and Carpenter, 1994). For example, if the availability of angling substitutes at comparable access costs in the landscape are limited, anglers might continue fishing on declining populations even if a perfect proportionality between population density and catch rates exists which may counteract a self-regulation (Post et al., 2002).

Selectivity

Recreational fishing is usually selective with respect to species, size classes, ages, sex, or behavioral traits. Some examples of these patterns are provided later.

Anglers prefer particular species either because they are considered of high culinary value, because they offer higher angling challenges, or because they are generally of greater appeal to the angler than other species. There is little room for generalization about species preferences of anglers. However, in many countries, large aquatic top-predators such as pike, pike-perch (*Sander lucioperca*), largemouth bass, brown trout (*Salmo trutta*), salmon (*Salmo salar*), or billfish species are the preferred target (e.g., Bogelius, 1998, Post et al., 2002; Arlinghaus and Mehner, 2004). In some recreational fishing, e.g., in the so-called match fishing in Europe, smaller zooplanktivorous or benthivorous species such as bream (*Abramis brama*), roach (*Rutilus rutilus*), or gudgeon (*Gobio gobio*) are the preferred species. As a general rule, however, in most recreational fisheries some components of the piscine food web receive a disproportionally higher fishing mortality than others. In some countries, almost any fish landed is removed for personal consumption, e.g., in some Eastern European countries, and anglers exploit those species that provide the greatest biomass per unit effort.

In addition, there are species-specific differences in the vulnerability to angling gear and catchability, such that angling may influence the distribution of nonnative and native species (Paul et al., 2003).

Angling is size-selective and targets larger, and often, but not always, older fish (Aas et al., 2000; Radomski, 2003). The size selectivity is mainly attributed to trophy fishing (Petering et al., 1995; Arlinghaus and Mehner, 2003; Isermann et al., 2005), but also to minimum length regulations which induce selective removal of the largest/oldest fish in the population (Arlinghaus et al., 2002a). In addition, depending on the target species, selective gear, fish morphology, as well as size-specific behavior and habitat preferences may contribute to a higher vulnerability of larger fish to angling (Paul et al., 2003; Schultz, 2004). There is some evidence from salmonids that individuals with high growth rates are more vulnerable to angling (e.g., Brauhn and Kincaid, 1982), which may be attributed to the dominance hierarchy and the greater opportunity of larger fishes to feed on bait (Tsuboi and Morita, 2004). On the other hand, as fish grow, the size of their prey often increases, thereby leading to the reduction of evacuation rate and feeding frequency. This in turn may result in lower vulnerability to angling as it has been shown for older walleyes (Serns and Kempinger, 1981; Newby et al., 2000; but see Isbel and Rawson (1989) for contradictory results). However, the exploitation rates of common angler species were often found to be much higher in large individuals (e.g., Pierce et al., 1995; Pierce and Cook, 2000).

Angling may also be selective towards certain sexes which may be attributed to behavioral sex-specific differences (e.g., Greenberg and Giller, 2001; Hutchings and Gerber, 2002). A higher catch of females was reported from longfin eel, *Anguilla dieffenbachii* (McCleave and Jellyman, 2004), pike (Casselman, 1975), carp, *Cyprinus carpio*, and salmon (Peréz et al., 2005). For walleye populations study results concerning the sex bias in the catch are inconsistent. Females may be caught more often if their growth rates are higher (Serns and Kempinger (1981) and literature therein) which can result in a higher mortality. In Chinook salmon, coho salmon, *O. kisutch* (Holtby et al., 1992) and white spotted charr, *Salvelinus leucomaenis* (Tsuboi and Morita, 2004) males were more vulnerable to angling. Guarding males of largemouth bass are particularly vulnerable to angling, because they show an aggressive behavior toward lures (Suski and Philipp, 2004).

Species or populations are often treated as fundamental units. However, individual fish within a population show individualistic behavioral traits (Greenberg and Giller, 2001). Consequently, they differ in their vulnerability to angling (e.g., Mezzera and Largiadèr, 2001). For example, individuals with a preference for littoral habitats or those showing a higher activity daytime might be more vulnerable to fishing if anglers also prefer fishing from the bank and during the day as opposed to boat or night fishing. Also domestication effects may influence the angling susceptibility of fish and consequently influence the distribution of stocked and native fishes (Mezzera and Largiadèr, 2001). In the context of size selectivity, also morphological aspects of the mouth, growth rate, metabolic rate, foraging, and dominance behavior—all of which is correlated with the individual size (Elliot, 1990)—are relevant traits angling may select for (Brauhn and Kincaid, 1982; Mezzera and Largiadèr, 2001; Garrett, 2002).

Consequences of High Exploitation Rates and Selectivity

Angling adds a further trophic level on aquatic ecosystems and anglers can be regarded as keystone predators in aquatic ecosystems (Hilborn and Walters, 1992). The fishing mortality can be rather high for particular, highly valued and sought after fish species (e.g., salmonids or percids) and within these species, larger size classes are positively selected. The combination of high exploitation rates and pronounced selectivity as summarized previously may have some direct and indirect effects on exploited fish populations, which will be discussed.

Direct Consequences

High fishing mortality, probably in combination with other exogenous stressors, has been shown to repeatedly influence fish population dynamics (Barker et al., 2002) and to contribute to the collapse of recreationally exploited fish populations (e.g., Lester et al., 2003; Sullivan, 2003; Wilberg et al., 2005). Post et al. (2002) documented examples from Canada wherein the recreational fishing led to a collapse in four Canadian high-profile inland fisheries (walleye, rainbow trout (*O. mykiss*), lake trout, pike). Four reasons may play a role and will thus be presented: depensatory mechanisms, truncation of age and size structure, loss of genetic variability, and evolutionary changes.

Depensation Instead of Compensation. Mechanisms of compensation are central to most classical fisheries biological concepts such as surplus production and maximum sustainable yield (MSY). It assumes that compensatory effects (e.g., enhanced growth rate, enhanced fecundity, enhanced juvenile survival) arise through attenuating intraspecific interactions and food competition when fishing reduces the abundance of the target population (Beard et al., 1997). The compensatory potential of fish populations, however, is a matter of debate (Rose et al., 2001). The relative strength and frequency of density-dependent population regulation depend on environmental conditions and life history strategies (Winemiller, 2004) and there is growing evidence that there are some limits for compensatory responses.

Some mechanisms, collectively referred to as depensatory responses, may counteract compensation if the population size is reduced below a specific threshold (Stephens and Sutherland, 1999; Lierman and Hilborn, 1997). After reaching such a threshold, group dynamics and cooperative interactions might be impaired, which compromise mating success (McCarthy, 1997), foraging (Day et al., 2001), or antipredator strategies (Rangeley and Kramer, 1998). Furthermore, large-bodied piscivores can exert a top-down control on smaller species that are competitors or predators of their own progeny (Walters and Kitchell, 2001). Reducing the abundance of large piscivores may relax smaller prey species from top-down control and impair the piscivores' potential for compensatory responses once the population is fished down under a threshold level. This might occur when the prey of the piscivores achieve a competitive advantage over the young of the piscivores. Also environmental stochasticity and genetic mechanisms such as drift and inbreeding may have a stronger influence on small populations and may impair their compensatory potential. Depensatory effects capture the positive relationship between the per capita population growth rate and population density at low population sizes, which increases the per capita mortality probability of intensively exploited fish populations at low population abundances. Depensatory mechanisms are known as the Allee effect in ecological literature (e.g., Stephens and Sutherland, 1999).

In aquatic systems characterized by relatively simple food chains, depensatory mechanisms can contribute to the failure of traditional management actions, prevent the rebound of populations after intensive recreational exploitation (Post el al., 2002), and greatly increase the likelihood of local and global extinctions (McCarthy, 1997). Especially predatory fish species (Duffy, 2002; Petchey et al., 2004) are sensitive against depensatory effects following high exploitation rates. Depensatory mechanisms may be important even in fish species characterized by a metapopulation structure, in particular in those living in fragmented habitats (Amarasekare, 1998). A metapopulation structure is found in migrating fish species that return to natal streams to spawn and is characterized by local populations linked by migration. It reflects a balance between local adaptation in the populations of river tributaries and a natural gene flow between those populations. The metapopulation structure is important for the abundance and diversity of most migrating salmonids and on the other hand, depends on natural gene flow and genetic variability within and between populations that may be affected by high exploitation rates (Policansky and Magnuson, 1998).

Truncation of Age and Size Structure. Size selective angling may not only reduce the biomass but also truncates the age and size distributions in the targeted fish population (Beard and Kampa, 1999; Radomski, 2003). Olson and Cunningham (1989) analyzed fishing contest records of common recreationally exploited fish species from Minnesota and observed a long-term decline in the numbers of large individuals of pike, muskellunge (*Esox masquinongy),* walleye, largemouth bass, bluegills, and black crappies. Shifts in the size and age structure towards higher percentages of younger and smaller fishes were observed in several exploited populations of rainbow trout and brown trout (Anderson and Nehring, 1984; Almodovar et al., 2002; Almodovar and Nicola, 2004), black crappies (Willis et al., 1994), smallmouth bass (*Micropterus dolomieui*) (Goedde and Coble 1981; Reed and Rabeni, 1989), walleye (Stone and Lott, 2002; Sullivan, 2003), lethrinids (Westera et al., 2003), yellow perch, pumkinseeds (*Lepomis gibbosus*), bluegills (Goedde and Coble 1981; Drake et al., 1997; Cook et al., 2001), pike (Goedde and Coble 1981; Pierce et al., 1995), and paddlefish (*Polyodon spathula*) (Scarnecchia et al., 1989).

The removal of large individuals may increase the growth rates of juvenile fishes if competition for food is relaxed at lowered population abundances (Ali et al., 2003). However, because the fish size correlates with many reproductive traits, the selective removal of most of the large individuals may probably affect the reproductive capacity of the exploited fish population despite compensatory growth of surviving individuals (Aday et al., 2002; Longhurst, 2002). Older fishes often have a higher hatching success than first-time spawners (Trippel, 1998) which may be attributed to a variety of factors such as egg size and quality or ideal spawning time. In many marine and freshwater fishes, many of them targeted by recreational fishery, age at maturity and egg size are positively correlated (Sargent et al., 1987) with larger age, size, or weight resulting in the production of larger eggs [black rockfish, *Sebastes melanops* (Bobko and Berkeley, 2004); grayling, *Thymallus thymallus* (Haugen and Vøllestad, 2000); haddock, *Melanogrammus aeglefinus* (Wright and Gibb, 2005); walleye and white sucker, *Catostomus commersoni* (Johnston, 1997); Atlantic salmon (Jonsson et al., 1996); brown trout (Olsen and Vøllestad, 2001, 2003); yellow perch (Lauer et al., 2005); striped bass, *Morone saxatilis* (Monteleone and Houde, 1990); cod, *Gadus morhua* (Trippel, 1998); pike (Wright and Shoesmith, 1988)]. The egg size, influenced by maternal effects (condition factor, weight, size or age at maturity, growth history, e.g., Johnston, 1997; Morita et al., 1999), but also genetically controlled (Su et al., 1997), positively correlates with offspring survival. For example, large salmonid eggs have higher survival rates than smaller eggs, when the concentration of dissolved oxygen is low (Einum et al., 2002). Vallin and Nissling (2000) pointed out that older individuals of Baltic cod produced larger eggs with neutral buoyancy which can be of crucial importance for egg and larval survival. In addition, egg size is often positively correlated with larval size and early growth (e.g., Wallace and Aasjord, 1984; Einum and Fleming, 2002) and both influence the probability of survival to maturity (Einum and Fleming, 2000). Larger larvae swim faster, avoid predation more easily (Miller et al., 1988), have higher feeding success (Marteinsdottir and Steinarsson, 1998), survive periods of low food supply for a longer time (Miller et al., 1988; Chambers and Leggett, 1996), and have lower winter mortality (Hurst and Conover, 1998). The age of the parental fish also influences the chemical composition of eggs, egg metabolism, and juvenile survival. The larvae of the oldest (17 years) rockfish females showed three times higher growth rates than larvae from the youngest females (5 years) (Berkeley et al., 2004a). With increasing age, the females provided more metabolic reserves (energy-rich triacylglycerol lipids) and these were positively correlated with subsequent growth and survival of the progeny.

Old and large fish also increase their reproduction success in breeding competition because higher competitive abilities enable them to obtain better spawning sites or—in case of salmonids—to dig deeper redds (Van den Berghe and Gross, 1984). Moreover, the fecundity in fish exponentially increases with age and size (Roff, 1983). Larger fish produce more eggs simply because of geometric constraints (e.g., Heyer et al., 2001; Palumbi, 2004), but also because they provide a greater proportion of energy stores to egg production (Berkeley et al., 2004a, 2004b).

The fish age also influences the spawning time. Younger and smaller fish may start later with spawning, because they emerge from the winter with lower lipid reserves than larger individuals and the need of acquiring sufficient energy reserves may delay spawning. The spawning time influences the recruitment, as the larval survival highly depends on the coincidence of larval production and peak zooplankton production (e.g., Plaza et al., 2004). An earlier birthdate may enhance the survival of the progeny presumably as a result of a longer growing season (Miranda and Muncy, 1987; Bobko and Berkeley, 2004). For fishes showing age-related temporal spawning, the removal of old age classes will shorten the spawning season and can result in recruitment failure in years when successful recruitment depends on early spawning time (Berkeley et al., 2004a). Because larval traits that confer a survival advantage likely vary according to the biotic and abiotic conditions in the lake, a diverse female age structure would be the best to protect a fish stock against variable environmental conditions (Heyer et al., 2001).

There are some additional positive effects of larger and older fish. Under natural conditions, large salmonids are more likely to survive migration and the period at sea. In the case of straying, large individuals can migrate longer distances and may contribute to an efficient gene flow within the metapopulation structure (Elliot, 1994). Furthermore, it has been shown for some fish species, that fishes like other vertebrates are capable of social learning from more experienced and sometimes older individuals concerning anti-predator behavior (Kelley and Magurran, 2003), migration and orientation (Odling-Smee and Braithwaite 2003), mate choice, foraging (Warburton, 2003), and communication behavior (for a review, see Brown and Laland, (2003). In addition, the recruitment in populations is not only influenced by cannibalism or intraspecific competition for food or space among fish of the same size. Older fish may also contribute to the regulation as it is assumed for pike and various salmonid species (Johnson, 1976; Mann, 1982; Elliot, 1994).

To sum up, there is much evidence that a natural and variable age structure improves recruitment and enhances the population's resilience to external disturbance (Heyer et al., 2001). In fish species exhibiting long life spans, the allocation of the reproductive output across many years can be seen as a bet-hedging strategy that ensures the reproductive success of at least some individuals under variable environmental conditions (Secor, 2000). Consequently, an estimation of recruitment may not be valid if age and size-specific differences in fecundity and the effect of maternal size and age on the progeny viability are not taken into account (Scott et al., 1999). In particular, already depleted populations may be negatively affected by the removal of adult and larger fish (Araguas et al., 2004).

Loss of Genetic Diversity. The genetic variability plays a crucial role in the survival of species (Brook et al., 2002; Frankham, 2003; Spielman et al., 2004) and is essential for their potential for successfully evolving in response to short- and long-term environmental changes (Ryman et al., 1995; Reusch et al., 2005). This aspect is of crucial importance especially in freshwater populations. Local populations of freshwater fishes are genetically more divergent than those of marine species (Ward et al., 1994) and are more susceptible to the loss of genetic variability (Ryman et al., 1995). In particular, small and isolated freshwater populations that are confronted with a high selective mortality combined with other threats, e.g., habitat loss or invasion of non-native species, are endangered (Stockwell et al., 2003).

Many fish species targeted by recreational fishers have a spatially phylogeographic structure defined by evolutionary history, demographic processes, the level of gene flow, and genetically based adaptations to the local environment which is detectable on different spatial scales (from river systems, tributaries, or even within rivers or lakes, e.g., Billington and Hebert, 1991; Bernatchez, 2001; Gerlach et al., 2001; Wirth and Bernatchez 2001; Nicod et al., 2004; Verspoor et al., 2005). The biodiversity on the level of discrete populations ensures their adaptive potential and the resilience against environmental changes of a species and plays a critical role in holding fisheries sustainable (e.g., Hilborn et al., 2003), particularly in fish species characterized by a metapopulation structure (Policansky and Magnuson, 1998). In particular populations living in an uncommon or variable habitat constitute an important part of the evolutionary legacy (Nielsen et al., 2001).

The reduction of population densities can lead to the loss of populations (e.g., Guinand et al., 2003) which obviously results in a loss of genes or gene combinations (Ryman et al., 1995). In addition, demographic bottlenecks are expected to reduce the genetic variation within a population (Hedrick and Miller, 1992) and its capacity for the retention of rare alleles (Hauser et al., 2002; Guinand et al., 2003) by genetic drift and inbreeding. A loss of genetic variation may further be caused by the skewing of the sex ratio as a result of the selective removal of male or female individuals from a population (Nelson and Soulé, 1987). In addition, the removal of the largest individuals may lower the genetic variability. Some genetic studies demonstrated a positive correlation between growth rate and protein/allozyme heterozygosity (e.g., Mitton and Grant, 1984; Borrell et al., 2004).

In general, the loss of genetic diversity and allelic richness decreases the adaptive potential and lowers the long-term fitness of populations (Lynch, 1991; Keller and Waller, 2002). In addition, there is some evidence that the genetic diversity on the level of individual organisms can provide fitness benefits (Britten, 1996; David, 1998; Wang et al., 2002; Reed and Frankham, 2003) and may increase disease resistance (Spielman et al., 2004). Until now, there are no studies available that have investigated these patterns in recreational fishing.

Evolutionary Changes Due to Selective Angling. Because many commercially exploited fish stocks declined and failed to recover even after ceasing exploitation (Hutchings, 2005), there is growing concern that heavy and selective exploitation over decades resulted not just in demographic consequences for targeted and nontargeted fish species, but may have led to detrimental evolutionary changes in some life history characters (Haugen and Vøllestad, 2001; Grift et al., 2003; Olsen et al., 2004). The changes of life history parameters in response to fishing are well known (e.g., Rijnsdorp, 1993), and the possibility that the fishery may inevitably change exploited fish stocks has been discussed for decades (Miller, 1957; Allendorf et al., 1987; Policansky, 1993). However, until now, the possibility of genotypic evolutionary changes received little attention in fisheries management, presumably because the difficulty to determine, whether the change of life history traits reflects phenotypic variability or is caused by genetic changes. However, the prerequisites for evolutionary changes in fish population in response to recreational fishing such as local adaptation, heritable population variation, and a high and selective fishing mortality (e.g., Linder et al., 1983; Gjerde and Schaeffer, 1989; Taylor, 1991; Stokes and Law, 2000) exist. The perception that evolution is a very slow process has been challenged by studies on fish species demonstrating that, under an appropriate life history and a sufficient strong selection pressure, a so called "contemporary evolution" (Stockwell et al., 2003) can occur in comparable short time periods, and change production-relevant life history traits such as age and size at maturation, growth rate, and annual reproductive investment (Reznick et al., 1997; Conover and Munch, 2002; Munch et al., 2005; Reznick and Ghalambor, 2005).

Only few studies on potential evolutionary changes as a consequence of high and selective fishing are concerned with angling. Diana (1983) observed a decrease in the age at first maturation in recreationally exploited pike (*Esox lucius*) populations. Magnan et al. (2005) examined the impacts of both recreational fishing mortality and interspecific competition on life history traits in 17 brook trout (*Salvelinus fontinalis*) populations. Both factors did not affect growth, but the age at maturity was inversely related to the intensity of recreational fishing and interspecific competition. Nuhfer and Alexander (1994) assumed that angling might have degraded the genetic growth potential of brook trout. Also the size structure and some reproductive traits of a bluegill population have been lowered by 30 years of intensive angling. The population exposed to high angling effort had younger ages and smaller lengths at maturation, and lakes exposed to a high angling effort had a higher proportion of cuckoldry in male bluegills as a result of the change in the size structure (Drake et al., 1997).

Also behavioral traits might undergo a selection in response to fishing (e.g., Heino and Godø, 2002). Behavioral individuality has a genetic basis (Skúlason et al., 1996; Iguchi et al., 2001) and some behavioral traits show heritabilities between 0.1 and 0.4. It has been demonstrated that angling creates a selection pressure against, for example, boldness (Mezzera and Largiadér, 2001) and for avoidance behavior (Garrett, 2002). Behavioral traits are usually determined by genes of more than one locus. Consequently, behavioral traits that are related to the vulnerability to angling can be correlated with other characteristics such as metabolic rates and parental care (Cooke, 2002). A selection against aggressive behavior may reduce the fitness of the surviving population. The aggressiveness of nesting male centrarchids correlated positively with the quantity of eggs in a male's nest. Consequently, the males with the greatest potential to contribute to annual recruitment were the most likely to be caught (and removed) by anglers (Suski and Philipp, 2004).

In addition, the preference for certain habitats or seasons by anglers may enhance the selection potential of angling because neither habitat choice nor migration time are necessarily random but may depend on sex (Mjølnerød et al., 1999; Hutchings and Gerber, 2002), size (e.g., Bremset and Berg, 1997), or genetic factors (Consuegra et al., 2005). For example, the seasonal exploitation of Atlantic salmon by anglers selectively removed individuals with certain phenotypic and genotypic traits and exerted an artificial selection on the time of the spawning migration. Because early and late runs of Atlantic salmon constitute genetically distinct components of the population, the selective angling may potentially disrupt the adaptive architecture of the population (Consuegra et al., 2005).

To sum up, angling may have the potential to cause an evolution in some life history traits (Policansky, 1993). Angling may select for or against certain life-history traits provided that the fishing mortality is high and the survivors represent genotypes that are less vulnerable to the force of mortality and then proliferate in subsequent generations (Conover, 2000). A prediction of the effects on life history evolution is difficult, because the effects depend on the multiple interactions within the aquatic ecosystem (Abrams, 1996; Gårdmark et al., 2003). However, changes in size-related, life history traits can influence population persistence and yield (Conover and Munch, 2002).

The higher the rate of fishing mortality and the higher the number of generations over those a population has been fished, the greater the probability that genetic responses occur (Hutchings, 2005). The outcomes of the selection are not necessarily positive, neither from the populations nor from the angler's point of view. Genotypes that survive fishing pressure may be less than optimal with respect to natural selection (Conover, 2000) and this may prevent a recovery of a population even after the fisheries have ceased.

Bycatch and Catch-and-Release Fishing. The negative impacts of bycatch are usually attributed to the commercial fisheries (Goñi, 1998; Harrington et al., 2005; Devine et al., 2006). But given the high amount of angling activity, bycatch can also be significant in recreational fisheries, if specific unwanted species, sizes, or sexes are partly or entirely released after the capture (Cooke and Cowx, 2004). There are also situations in which the angler has the predetermined aim to release the fish that are caught. In such a situation, bycatch is a misleading term, but catch-and-release still occurs with potential impacts on the fish released alive. In particular, specialized anglers practice voluntarily catch-and-release (C&R) fishing (e.g., Sutton and Ditton, 2001; Policansky, 2002; Millard et al., 2003), and the C&R rates are increasing worldwide (Bartholomew and Bohnsack, 2005).

Fish are also released after capture because they are protected by minimum size limits or creel limits (Millard et al., 2003). Also management rules or legislations demand the release of all captured fish or of fishes of protected size classes or species (Orciari and Leonard, 1990; Anderson et al., 1998; Bartholomew and Bohnsack, 2005).

However, even if many fishes are released that might ameliorate the downsides of intensive recreational fishing, there may be negative effects of C&R on fish populations (Cooke et al., 2002). Depending on species, populations, or other fish-related factors such as stage of migration, degree of starvation (Brobbel et al., 1996), or fish size (Ferguson et al., 1993; Wilde, 1998; Davis and Parker, 2004), C&R can cause immediate or delayed post-release mortality (Hartley and Moring, 1995; Edwards et al., 2004). The mortality of ictaluridae ranged from 0% to 33%, of esocidae from 0% to 30%, of salmonidae from 0% to 82%, of percidae from 0% to 40%, of centrarchidae from 0% to 88%, of percichthyidae from 0% to 70%, and of sciaenidae from 0% to 70% (see Beggs et al., 1980; Muoneke and Childress, 1994; Lindsay et al., 2004; for more species, see Bartholomew and Bohnsack (2005)). The most important mortality factors are the hooking location and the injuries caused by fishing gear and handling, but also environmental conditions (water temperature, catch depth), type of fishing gear, fishing method, and angler experience (see Bartholomew and Bohnsack (2005) for review).

Deep hooking (hooking in the gills, esophagus, intestine, eyes) causes a high mortality and decreases growth rates, especially if accompanied by bleeding (e.g., Aalbers et al., 2004; Meka, 2004), whereas hooking in the lips or jaw area consistently causes minimal mortality, which is consistently less than 5% and often less than 1% (Lindsay et al., 2004). The use of landing nets compared with a landing by hand increased the mortality. Landing nets caused fin abrasions, scale and mucus loss, with knotted mesh types being more injurious than rubber or knotless nets (Barthel et al., 2003).

Nonlethal injuries may expose the fish to parasites or bacterial and fungal infections (e.g., Steeger et al., 1994), or reduce its ability to forage effectively or to escape predators (Cooke et al., 2003a, 2003b). The location of the wound site has been demonstrated to be a function of hook size and type, bait or lure type, and additional situational factors. In general, barbed hooks caused more and more severe hooking injuries and increased the handling time (Muoneke and Childress, 1994; Cooke et al., 2001a; Lindsay et al., 2004; but see Schill and Scarpella, (1997)) and circle hooks caused less damage to the fish (Schaeffer and Hoffman, 2002; Cooke and Suski, 2004). Angling with artificial flies or lures resulted in a lower mortality in salmonids than angling with natural baits (Taylor and White, 1992). However, various species may respond differently, because of species-specific foraging behavior or mouth morphology (Cooke et al., 2003a, 2003b).

Catch and handling may cause a similar stress response in the caught fishes as the visual or chemical contact to piscivorous predators (Cooke et al., 2003c). In addition, injuries following the catch may lead to the release of infochemicals associated with the predation (e.g., so-called Schreckstoffe) which may cause stress responses in conspecies (Rehnberg et al., 1987). Although there are species-specific differences in the stress response (e.g., Mazeaud and Mazeaud, 1981), the stress response of fishes is generally similar to other vertebrates (Wendelaar Bonga, 1997). The physiological response of a prey associated with predation (cardiovascular disturbance, release of stress hormones, e.g., Cooke et al. (2003b)) may lead to an alteration in physiology and behavior. Such nonlethal costs (Lima and Dill, 1990) can reduce the potential for survival by diminishing the capability to deal with basic ecological challenges such as foraging and predator avoidance (Martel and Dill, 1993; Allouche and Gaudin, 2001) and can negatively affect immune system (e.g., Davis et al., 2003), growth (e.g., Small, 2004), and reproductive success (e.g., Schreck et al., 2001; Ostrand et al., 2004). Cooke et al. (2000) reported that C&R reduced the activity level of nest guarding behavior in black bass (*Micropterus* sp.). Angling and release increased the nest abandonment (Philipp et al., 1997) and reduced the ability of angled and released males to defend their brood (Suski et al., 2003). The short time removal of nest guarding smallmouth bass (*Micropterus dolomieu*) led to an increase in predation by round goby (*Neogobius melanostomus*) which entered abandoned nests (Steinhart et al., 2004). However, many studies show that fishes can quickly recover from stress (Schreer et al., 2001; Cooke et al., 2003d) as long as the detrimental effects were minimized.

Tournament angling is an important part of fishing activities in many countries (e.g., Schramm et al., 1991). It is often associated with C&R fishing. However, compared to noncompetitive C&R fishing, fishes caught by tournament anglers may suffer higher mortality due to prolonged retention time (Neal and Lopez-Clayton, 2001; Graeb et al., 2005), bad water quality (Carmicheal et al., 1984; Furimsky et al., 2003), extensive handling, and additional stress (Hartley and Moring, 1995; Killen et al., 2003; Edwards et al., 2004). In particular, the weigh-in procedures can cause significant stress for the fish (e.g., Suski et al., 2004).

Indirect Consequences

Effects on Trophic Cascades. The fish species targeted by angling can be a predator or a competitor or can influence habitat structure und environmental conditions. Depending on role and dominance within the ecosystem, the removal of a substantial part of a population can significantly affect the trophic process and the community structure (De Roos and Persson, 2002; Donald and Anderson, 2003), especially in aquatic ecosystems where the strength of the top-down control is greater than in terrestrial ecosystems (Shurin et al., 2002) and the natural diversity at higher trophic levels is low (Petchey et al., 2004). According to the trophic cascade theory, abundance, and species composition at the top predator level affect the lower trophic levels of aquatic ecosystems via the top-down control of the community structure (Carpenter et al., 1985; Duffy, 2002). Changes in the abundance of top predators control a cascade of trophic interactions that regulate zooplankton community structure, algal dynamics, and nutrient cycles in marine and freshwater ecosystems (e.g., Brett and Goldman, 1996; Pace et al., 1999; Reid et al., 2000; Findlay et al., 2005). Consequently, the sequential removal of keystone species can change the structure of an ecosystem (Ward and Myers, 2005; Worm et al., 2005) undermine its resilience (Scheffer et al., 2001; Hughes et al., 2005), and change the biogeochemical processes between the ecosystem and the atmosphere (Schindler et al., 1997)—effects that may be difficult to reverse (Paine et al., 1985). Although a controversial issue in the recreational fisheries management (Cowx, 2002b), the predator control is an important management tool, for example, in the context of biomanipulation. While there are many studies on biomanipulation (e.g., Perrow et al., 1997; Carpenter and Lathrop, 1999; Mehner et al., 2004), the consequences of species-selective angling are rarely studied. Pinnegar et al. (2000) concluded from a literature review that artisanal and recreational fishery in littoral habitats may have the potential to induce cascading effects. The structure of aquatic food webs and the influence of environmental factors may facilitate compensatory mechanisms, which complicate the prediction of the outcome of changes on a top predator level (Ruetz et al., 2002; Steiner, 2003; Turner, 2004). However, given the species preference of recreational fishers and the high harvest rates it can be assumed that recreational fishing also has the potential to affect the trophic structure and thereby alter aquatic ecosystems.

Trait-Mediated Effects. Due to the adaptive flexibility in prey behavior, morphology, physiology, and ecology in response to variable predation risks, a change in the abundance of predators has ecological consequences independent of the lethal effects on a prey species (Wootton, 2002; Dill et al., 2003; Werner and Peacor, 2003). Anglers acting as predators may play a role in trait-mediated interactions. If animals avoid a refuge area as a result of disturbance or predation (Frid and Dill, 2002), these animals may experience greater predation by other predators, e.g., by avian predators (Crowder et al., 1997; Sih et al., 1998; De Goeij et al., 2001) or experience greater competition in suboptimal or crowded habitats. In addition, the avoidance of certain areas may locally influence their predators and may on the other hand lead to an increase in their prey (Dill et al., 2003) and to a decline of their prey in the refuge habitats (Bernot and Turner, 2001; Lenihan et al., 2001). These "trait mediated effects" (Abrams, 1995) can be important in structuring food web interactions (Biro et al., 2003; Schmitz et al., 2004), but have not been investigated in a recreational fishing setting so far.

Effects Resulting from the Angling Activity

Disturbance of Habitats

Recreational fishermen have access to nearshore and littoral habitats which are of crucial importance for many fish species (Schiemer et al., 1995). As interfaces between terrestrial and aquatic ecosystems, littoral zones fulfill a variety of physical and ecological functions, delay or prevent the transport of nutrients to lakes from eroding upland soils (Carpenter and Lathrop, 1999), enhance local energy and nutrient availability fostering a higher biological productivity, support material and energy cycles and a variety of life history strategies (Naiman and Decamps, 1997). The diverse riparian zone processes affect biodiversity, reproduction, feeding, and predator-prey interactions (Pusey and Arthington, 2003). Woody debris and submerged or emerged macrophytes are refuge or feeding habitats for juvenile fish and invertebrates (Persson and Eklöv, 1995; Lewin et al., 2004), and macrophytes serve as spawning substrate for phytophilic species (see Aarts and Nienhuis, 2003). At the same time, the alteration of littoral habitats by human activities has resulted in a loss of refuge habitats and resource heterogeneity (Ostendorp et al., 1995; Radomski and Goeman, 2001) which in turn affected fish communities by changing the fish species richness (Trial et al., 2001), biomass (Radomski and Goeman, 2001), growth rates (Schindler et al., 2000), and the spatial distribution of fish (Scheuerell and Schindler, 2004).

Anglers can affect littoral habitats if they make paths to gain access to the water and walk parallel to the shoreline. A medium or heavy use of pathways and shores can change or destroy the natural plant communities of freshwater or marine littoral habitats (Rees and Tivy, 1978; Brosnan and Crumrine, 1994; Sudmann et al., 1996). A single wading on shallow, salmonid-spawning habitats during the period before the hatching killed 43% of eggs and fry, while a twice-daily wading killed up to 96% (Roberts and White, 1992). Anglers may also cut bank vegetation and remove submerged vegetation at the beginning of the fishing season. The removal of the aquatic plants and shoreline vegetation can affect phytoplankton development, invertebrates, fishes and birds (Brooker and Edwards, 1975; Sukopp and Markstein, 1981), enhance erosion processes (Williams and Moss, 2001 and literature therein), and can change nutrient fluxes (Liddle and Scorgie, 1980). Compared with other recreational activities, the effects of angling on the aquatic vegetation may be of minor importance (Sukopp, 1971). However, the impacts vary according to dimension and habitat involved. Overall, the disturbance and destruction of benthic habitats, often attributed to the commercial marine fisheries (Kaiser et al., 2000), seems to be less important in recreational fishing.

Disturbance of Wildlife

Nearly all activities carried out on the shores are potentially disturbing wildlife, living in littoral areas or sitting on the surface. The disturbances associated with recreational fishing originate mainly from direct contact, sound, and sight. In general, antipredator behavior has been evolved to generalized threatening stimuli such as noise or fast-approaching objects. It creates a trade-off between the avoidance of a predation risk and other fitness-related activities (Frid and Dill (2002) and literature therein). The predation risks may increase vigilance (Lima and Bednekoff, 1999) and induce, depending on the prevailing conditions (Gill et al., 2001), habitat shifts (e.g., Werner and Hall, 1988). Safe habitats may be less profitable and animals may experience a greater predation risk when searching additional food in other, more dangerous, habitats to avoid starvation

(McNamara and Houston, 1987). Also the reproduction (see Magnhagen (1991) for review) and the parental investment (compare Mueller, 1980) may be negatively affected. By influencing resource allocation, reproduction, or the spatial distribution of individuals, disturbances are assumed to indirectly affect not just an individual's fitness but also the population dynamics.

Above all, water birds are closely associated with littoral and shoreline habitats. Therefore, the research on human disturbance of wildlife in aquatic ecosystems concentrates mainly on water birds. Quan et al. (2002) demonstrated that species richness and abundance on a highly exploited lake were correlated to human disturbance and not to habitat quality. Human disturbances, especially those caused by recreational activities, can affect distribution, species richness, and abundance of waterbirds (Robinson and Cranswick, 2003) by disturbing overwintering (Bell and Austin, 1985), resting and feeding (Madsen, 1985; Mori et al., 2001), and reproduction (e.g., prelaying phase, egg and chick phase; e.g., Gillett et al., 1975; Parsons and Burger, 1982). The disturbance of feeding may be more pronounced if the feeding is restricted to certain places or time periods and can result in adults having insufficient time to fulfill their own energy demands and those of their chicks (Leseberg et al., 2000). The disturbances to the nesting birds can result in higher rates of nonhatching and abandonment (Safina and Burger, 1983; Piatt et al., 1990), in the exposure of eggs to predators (Kury and Gochfeld, 1975), or unfavorable environmental conditions such as solar radiation (Hunt, 1972) or thermal stress (Tremblay and Ellison, 1979) and may therefore decrease breeding success. The effects of human disturbances may increase if breeding birds that have lost their eggs or chicks predate on conspecifics (Hand, 1980). Compared to other land-based activities such as bird-watching, walking, or picnicking, shore angling is considered to have serious impacts on water birds, since anglers often use vehicles to gain access to the angling sites (Watson et al., 1996) and remain there for long periods. Furthermore, they frequently show long periods of inactivity interspersed with short periods of rapid movements (Bell and Austin, 1985). Liddle and Scorgie (1980) cite some studies which showed that activities by anglers substantially decreased the breeding success and breeding stocks of different water bird species. Anglers had a similar effect as boats on water birds, creating an area around them within which birds did not venture. According to this, Sudmann et al. (1996) observed a reproduction failure of breeding waterbirds in a nature reservoir during years angling took place. The reproduction improved after termination of angling.

The avoidance and redistribution in response to human disturbances are species-specific (e.g., Mori et al., 2001; Rodgers and Schwikert, 2002; Stolen, 2003). Species that do not avoid disturbance may be affected by disturbance even more seriously, if they are forced to tolerate the disturbance in case suitable alternative habitats are lacking (Gill et al., 2001). Other bird species may show an adaptation to recreational disturbances. The great crested grebes (*Podiceps cristatus*) left their nest at shorter distances to approaching rowing boats presumably as an adaptation to recreational activities. However, the short flight distances were disadvantageous, as the birds did not cover their eggs before leaving, so that the clutch was not protected from predation (Keller, 1989).

Few studies deal with the angling associated indirect disturbance of other taxa than birds. Angling may disturb otter (*Lutra lutra*) populations if there is a lack of sufficient refuges such as dense woodland structures along river banks, especially if anglers prefer remaining cut-back trees and stumps as fishing sites. The removal of woody structures can be connected with measures to improve angling opportunities (MacDonald et al., 1978) and fisheries resources. However, the debrushing and removal of woody structures is a controversial issue and there are some results that the placement of woody structures can improve the abundance and size of salmonid fishes (Angermeier and Karr, 1984; Zika and

Peter, 2002). Furthermore, the species composition of dragonflies can reflect disturbances of littoral vegetation caused by recreational fishermen. The numbers of rare species was negatively correlated with the degree of degradation of shoreline vegetation (Müller et al., 2003).

Boat Traffic

Wolter and Arlinghaus (2003) categorized the effects of boat traffic on fish, most notably on fish larvae, into direct and indirect stressors. Direct effects were caused by the physical forces generated from moving ships, directly related to fish mortality (propeller action, waves, wash waves, and dewatering). Indirect effects results from stress, disturbances preventing fish from feeding or nest-guarding, dislodgement of eggs or larvae, an increase of turbidity, or a loss of macrophyte following wave actions. Their review deals mainly with the impacts of commercial navigation in waterways and some effects may be restricted to large vessels. However, wave effects may also result from recreational boat traffic (Arlinghaus et al., 2002b; Maynord, 2005).

Experimental studies have shown that shear stress can increase the mortality of eggs and larvae from different fish species (Morgan et al., 1976; Killgore et al., 1987). Given the smaller size of the boats used by recreational fishermen, impacts of shear stress, stranding, and dewatering following wave action may be less important (Arlinghaus et al., 2002b). But obstructing nest-guarding behavior and dislodgement and redistribution of eggs and larvae may affect the fitness of fish populations. Small recreational boats travelling at slow speed near the nests drove males of longear sunfish (*Lepomis megalotis*) from the nests, thus increasing the likelihood of egg predation. Boats moving at higher speeds increased the turbidity and therefore the possibility of the predation success (Mueller, 1980). The passage of even a single paddle or motorboat over low-density organic mud can lead to a resuspension of sediments and if the frequency of boat traffic is sufficient during the season, it may result in an increase of turbidity (Garrad and Hey, 1987). In shallow lakes, water mixing by small motorboats increased the turbidity and the ortho and total phosphorus concentrations in the water depending on bottom sediment (Yousef et al., 1980). An increase in turbidity beyond the natural level may have physiological effects (gill trauma, Berg and Northcote (1985); sublethal stress response (Redding et al., 1987)) and behavioral effects (avoidance, predator-prey interactions, e.g., Barrett et al., 1992; Horppila et al., 2004). Additionally, the increased turbidity may contribute to a loss of macrophytes in littoral habitats and macrophytes serve as colonization substrate for various prey organisms, as spawning substrate for phytophilic species and as feeding and refuge habitat for juvenile fish.

Motorboat traffic in rivers, lakes, and along the coastline results in the emission of inorganic and organic compounds into the water and into the air near the surface, which are toxic to zooplankton (Jüttner et al., 1995) and fishes (Tjärnlund et al., 1995). Also in marine ecosystems, the engine emissions from outboard motors can contribute to the surface microlayer, and the toxic substances on the air-water interface can significantly affect the survival and development of early life history stages of marine fishes and other surface-dwelling organisms (Hardy et al., 1987; Kocan et al., 1987).

Even if it is not possible to quantify the effects of boat traffic linked exclusively to angling, given a substantial level of boating activity, there could be some negative effects on the aquatic environment or fish stocks, whereas the effects depend on motor type, traveling speed, bottom structure of the ecosystem, or slope of the shoreline.

Noise

Noise is usually not linked to angling, but may be attributed to boating during angling. Fish obtain information about predators, prey, competitors, and potential mates by listening to their acoustic environment. The loss of hearing abilities could decrease the distances over which such information could be obtained, and deteriorate the quality of the obtained information (Popper, 2003, Popper et al., 2004, and literature therein). Consequently, the acoustic communication may be crucial for the survival of fish species. A fish's sensitivity to noise depends on the species (e.g., Kenyon et al., 1998), e.g., on the presence of a gasbladder or other gas-holding structures, and the coupling or proximity of these structures to the inner ear.

Although sound is an important means of communication in aquatic environments, the effect of anthropogenic noise on fishes is poorly understood (Smith et al., 2004a). However, loud sounds elicited flight responses in herring (*Clupea harengus*), cod, and haddock (e.g., Schwarz and Greer, 1984; Engås et al., 1996). Experiments showed that fish are susceptible to noise-induced stress and hearing loss (Scholik and Yan, 2002a). Noise exposure caused short-term stress responses (release of catecholamines followed by the activation of the stimulation and secretion of glucocorticoid homons) and decreases in auditory thresholds in goldfish, *Carassius auratus* (Smith et al., 2004a), and effects of the auditory sensitivity were further shown in catfish, *Pimelodus pictus* (Amoser and Ladich, 2003), bluegill (Scholik and Yan, 2002b), tilapia, *Oreochromis niloticus* (Smith et al., 2004b), and fathead minnow, *Pimephales promelas* (Scholik and Yan, 2002c). Noise from small boats (55 hp outboard engines) significantly elevated auditory threshold of fathead minnows and led to significant changes in hearing capability (Scholik and Yan, 2002c). Boat noise, recorded in the field, caused stress in the hearing specialists gudgeon and carp, as well as in the hearing generalist Eurasian perch (*Perca fluviatilis*) (Wysocki et al., 2006)

Although the physiological and auditory injury may not be permanent, the sound may mask the detection of biologically relevant signals and induce a stress reaction, which may have deleterious effects on the fish population.

Loss of Fishing Gear

The debris connected to recreational fishing consists of lost fishing lines, lead sinkers, and general litter. Studies estimating the quantity of angling litter showed that the dimension of angling litter can locally be rather high (Bell et al., 1985; Forbes, 1986; Cryer et al., 1987a; Chiappone et al., 2004). Lost hook-and-line gear can have a negative impact on health and survival of sessile invertebrates (Asoh et al., 2004) and may be dangerous for organisms such as water birds or turtles (Chiappone et al., 2005). In contrast to general litter (e.g., plastic bags, paper), which mainly causes visual disturbances, nylon fishing lines are potentially hazardous to water birds (Cowx, 2002a). The discarded lines can ensnarl waterbirds, resulting in death by starvation, drowning, or suffocation (Bell et al., 1985).

Elevated lead exposure resulting from metal mining, traffic, and ingestion has been recognized to cause a substantial mortality particularly in water birds. One of the earliest published reports on lead poisoning of water birds is the study of Bellrose (1959) on the effects of the ingestion of lead from shotgun pellets on water birds. Also the lead sinkers used by anglers may cause a serious threat to wildlife. Angling represents a proportion of up to 14% of the total amount of lead annually discharged into the Canadian environment. The annual loss of lead sinkers and jigs is estimated to introduce about 500 t of lead into Canadian water bodies (Scheuhammer et al., 2003), and up to 200 t into Swedish river mouth areas (Jacks et al., 2001). Lead has a relatively high intrinsic toxicity. Similar to other heavy metals, lead is a stable and persistent environmental contaminant since it cannot be degraded easily. Therefore, lead tends to accumulate in the environment. Lead sinkers in Swedish rivers showed slow dissolution rates of approximately 1%, provided the sinker was not buried in the sediment (Jacks et al., 2001). The authors assumed that the dissolved lead ends up in the sediment of the downstream waters with possible effects, depending on lead concentrations and on local ecosystems. Some studies showed that lead contamination is a significant mortality factor in different swan species or subspecies (Mudge, 1983; Spray and Milne, 1988; for review, see Blus (1994)). Water birds ingest lead sinkers because they mistake small lead sinkers for food items while fish-eating birds ingest lead sinkers when they consume lost bait fish, with hook and weight still attached and is assumed to be the only significant source of elevated lead exposure in loons (Scheuhammer et al. (2003) and literature therein). Lead poisoning through anglers' lead sinkers affected swan populations (*Cygnus olor*) via direct mortality and sublethal effects such as altering the blood characteristics and lowering reproductive success (Simpson et al., 1979; Sears, 1988). Other bird species such as loons, merganser, pelicans, cormorants, and gulls were also reported to have died after the ingestion of lead sinkers (Twiss and Thomas, 1998; Franson et al., 2003; Scheuhammer et al., 2003). The number of poisoned birds may be underestimated, because lead poisoning is a chronic disease and affected birds can be expected to seek cover before dying (Franson et al., 2003). Because of the high environmental persistence, lead can accumulate in the environment and can be transported in the food chain to species of higher trophic levels (Garcia-Fernandez et al., 1995). For example, raptorial birds can suffer from secondary lead poisoning after eating waterfowl or lost bait fish with the line and sinker still attached (Clark and Scheuhammer, 2003). While most secondary contamination of predators derives from ingestion of lead shot and some lead pellets, and pieces from lead sinkers may pass harmlessly through the digestive tract of predators or are regurgitated, there is also a risk of poisoning from lead fishing sinkers. Additionally, there are also cases of human infants being poisoned by accidental ingested lead sinkers (e.g., Mowad et al., 1998).

Bait Digging

With the popularity of recreational fishing, the demands for live bait rise. Some studies on marine coastal habitats have shown that bait digging can locally influence the littoral fauna (Beukema, 1995; Beukema et al., 2002) and that the bait collection affected the abundance and size structure of harvested benthic organisms (e.g., Cryer et al., 1987b; Keough et al., 1993; Roy et al., 2003). Some of the species intensively used as baits can play an important role in structuring the bottom communities. Therefore, an intensive harvest affects not only the harvested species but other components of the macro- and meio-fauna, as well as bacteria and algae and the meio-fauna. For example, such cascading effects can result from an intensive collection of sandprawns, *Callianassa kraussi* (Wynberg and Branch, 1997; compare also Van den Heiligenberg (1987)). Ghost shrimps, *Trypaea australiensis*, a popular bait species in Australia on tidal flats and beaches, are often collected by either digging or using hand-operated suction pumps (Contessa and Bird, 2004) and their harvest changed the distribution and abundance of other benthic taxa (polychaetes, amphipods, soldier crabs). In addition, the reduction of benthic organism (cockles or worms) may potentially affect the behavior and foraging success of higher trophic level species such as shorebirds (Shepherd and Boates, 1999).

The bait digging or pumping and the associated trampling can involve a considerable disturbance to the sediment and affect taxa, sensitive to disturbance of the sediment structure (Brown and Wilson, 1997; Wynberg and Branch, 1997). There is some evidence that the intensive bait digging for lugworm (*Arenicola marina*) and ragworm (*Nereis diversicolor*) reduced the abundance of cockles (*Cerastoderma edule*). The digging can lead to a burial of many cockles and to a surface exposure of some others. A burial below 10 cm prevent their return to their normal position in the surface layer (Jackson and James, 1979) and animals lying on the surface were exposed to predators (Wynberg and Branch, 1997; Skilleter et al., 2005).

In addition, bait collecting can affect not only the biological but also the physical and chemical sediment parameters. Bait pumping and trampling changed porosity, organic carbon content, and redox potential of the sediment and increased the chlorophyll concentration (Contessa and Bird, 2004). There are also indications that the perturbation of the sediment through intensive digging influenced bioavailability and uptake of heavy metals (lead and cadmium) by polychaetes (Howell, 1985).

Rocky coastal communities can also be affected by human activities (Castilla, 1999). The intertidal and subtidal boulders on rocky shores exhibit a diverse assemblage of sessile and mobile fauna. Consequently, a frequent sampling can cause short term effects on the abundances of sessile organisms (Chapman and Underwood, 1996). For example, the collection of mussels used as baits by anglers significantly reduced cover, density, biomass, and size of the mussels (*Mytilus californianus*) on rocky shores even during a period of high natural disturbance (Smith and Murray, 2005). The population dynamics of cunjevoi (*Pyura stolonifera*) has also been affected by bait collection (Fairweather, 1991).

Nutrient Input

Groundbaiting or chumming is widely practiced in freshwater by some anglers to attract fish such as the cyprinids bream, carp, or tench (*Tinca tinca*) to the angling site (Arlinghaus and Mehner, 2003). Ground baiting up to a certain limit can be effective in increasing the carrying capacity of the fishery and the catch of cyprinid fishes (Arlinghaus and Niesar, 2005). Higher groundbaiting rates can negatively affect the catch (Wolos et al., 1992). The existence of an upper limit may result from negative impact of not consumed baits on the water quality and invertebrate community. Groundbaiting over the entire fishing season may lead to significant changes in the benthic invertebrate community. The rapid breakdown of cereal baits by microbial activity resulted in a high oxygen consumption of the sediment (Cryer and Edwards, 1987). Presumably, as a result of the alteration of the microbial and chemical conditions through the decay of uneaten baits, the densities of naididae, cyclopoidae, and cladocera decreased. Only tubificidae showed no reductions in density (Cryer and Edwards, 1987). In general, the lack of oxygen on the sediment surface can result in diminished decomposition rates, causing an accumulation of organic surplus. The decay of this organic matter can enhance the ammonium flux from the sediment and initiate the redox-dependent release of iron-bound phosphorus, therefore contributing to the internal nutrient loading (e.g., Søndergaard et al., 2001). At the same time, nutrients especially phosphorus from the egestion or excretion of fish after having fed these baits or from uneaten baits may substantially contribute to anthropogenic eutrophication and therefore either directly or indirectly enhance primary production and algal growth (Arlinghaus and Mehner, 2003; Niesar et al., 2004). On the other hand, the angler harvest can counterbalance the nutrient input from groundbaiting (Wolos et al., 1992; Niesar et al., 2004). However, such a harvest rate may be unrealistic as many specialized anglers mainly practice C&R fishing (Sutton and Ditton, 2001; North, 2002; Arlinghaus and Mehner, 2003). However, the contribution of groundbaiting to an anthropogenic eutrophication is strongly dependent on local conditions. Water depth, trophic state, effective nutrient load and loading history, water retention time, as well as fisheries-connected factors (harvest rates, digestability, and nutrient composition) affect the impact of the groundbaiting on the water body. Small, shallow, oligotrophic lakes with long water retention times, high angler densities, and low harvest rates may be sensitive to ground baiting (Arlinghaus and Mehner, 2003). Until now, the input of other chemicals such as preservatives that may leach from commercial baits received no scientific attention.

Fish Introductions and Stocking. This review focuses on the effects of exploitation and associated disturbances on fish populations. However, it is necessary to mention that a major impact of the fishery on aquatic ecosystems originates from deliberate or accidental fish introductions and fish stockings (Krueger and May, 1991; Grimes, 1998). Depending on the ecology of the introduced species and the environmental conditions, the introduction of nonindigenous fish can have considerable long-term and widespread biological effects on native fish stocks. Introduced fish have threatened native species via competition, predation, and hybridization (e.g., Selgeby, 1998; French and Jude, 2001; Novinger and Rahel, 2003; Simon and Townsend, 2003; Weber and Fausch, 2003, for review; Fritts and Pearson, 2004; Winfield and Durie, 2004; Holthe et al., 2005), via the spread of diseases and parasites (e.g., Hedrick et al., 2000; Lafferty et al., 2004; Gozlan et al., 2005) and have caused severe economic losses (Pimentel et al., 2000).

The biological effects of stocking largeley depend on the origin of the stocked fish. In particular the introduction of alien species can have disastrous effects on the native fish communities and ecosystems (e.g., Parker et al., 2001; Perry et al., 2002; Kats and Ferrer, 2003; Simon and Townsend, 2003), and may contribute to the homogenization of the worldwide freshwater fauna (Rahel, 2002). Consequently, the legislation in many countries restricts or prohibits the introduction of non-native species.

Also the stocking of fish species native to the country or region, however, can have detrimental effects on the ecosystem if it biases the natural community structure. It can affect abundance, size distribution, or behavior of other vertebrate or invertebrate taxa (Barthelmes et al., 1998) as well as modify environmental conditions (Ten Winkel and Meulemans, 1984; Lougheed et al., 1998), trophic structure of ecosystems (e.g., McNaught et al., 1999), and nutrient fluxes (Elser et al., 2000; Attayde and Hansson, 2001; Schindler et al., 2001).

As mentioned above, many fish species are characterized by an adaptation to the local environment. Translocations of the same fish species across watersheds can destroy the phylogeographic structure of fish species. The disruption of the local adaptation by hybridization, introgression of genetic material from the stocked into wild population and outbreeding may affect the fitness of the local populations (e.g., Cooke et al., 2001b; Hansen et al., 2001; Poteaux et al., 2001; Araguas et al., 2004; Gilk et al., 2004; Granath et al., 2004). If the introduced fishes build up self-sustaining populations, they can contribute to the homogenization of worldwide fish fauna (Rahel, 2002).

The effects may be especially deleterious if stocked and wild fish differ in origin and genetic diversity or behavioral, morphological, and physiological traits resulting, for example, from selection differences between hatchery and natural environments as it has been demonstrated by many researchers (e.g., Einum and Fleming, 1997; 2001). Many studies demonstrated a loss of genetic variance between and within populations (e.g., Reisenbichler et al., 1992) and the occurrence of outbreeding depression (Cooke et al., 2001b; Gilk et al., 2004; Granath et al., 2004) following the hybridization between fish of different origins. In addition, stocking has often been considered unsuccessful because captive reared fish performed poorly under natural conditions (e.g., Jonsson, 1997; Miller et al., 2004) primarily due to inappropriate anti-predator (Berejikian, 1995; Alvarez and Nicieza, 2003), foraging (Reiriz et al., 1998; Munakata et al., 2000), or reproductive success (Fleming et al., 1996; Berejikian et al., 1997) as a consequence of hatchery shortfalls such as domestication, genetic drift, or inbreeding (Waples, 1999; Brown and Day, 2002; Glover et al., 2004), even if individuals from the wild population were continuously introduced in the breeding stock (Lynch and O'Hely, 2001).

The transfer of live bait fish via anglers have led to some introductions of nonindigenous species in different water basins in Europe and the U.S. (Ludwig and Leitch, 1996; Winfield and Durie, 2004; Holthe et al., 2005). The high numbers of anglers, their mobility and baitfish maintenance technology increase the probability that angler distribute live bait fish between different water basins (Ludwig and Leitch, 1996). Live baiting has been found to include many different species and there are some indications that some were able to establish self-sustaining populations (Adams and Maitland, 1998; Winfield and Durie, 2004).

Besides the risks of unintended introductions, live and even frozen bait fish from both the wild and the bait fish industries have been shown to transfer different viruses which can seriously affect the health of wild fish populations (Goodwin et al., 2004). However, it is worth mentioning that a lot of regional regulations prohibit the transport and release of live bait fish, or in some cases, even the use of live bait fish.

Implications for Management and Conservation

It is increasingly accepted that uncontrolled angler effort may likely result in a collapse of the less productive fish populations in particular (Cox and Walters, 2002). The increased awareness that aquatic resources are not infinite (FAO, 1996; Cowx and Van Zyll de Jong, 2004) led to the view that sustainable fisheries management is globally needed in both commercial and recreational fisheries (Garcia and Staples, 2000; Arlinghaus et al., 2002a). Such managment allows a long-term persistence of aquatic ecosystems while balancing human requirements against the protection of aquatic environment and biodiversity (Arlinghaus et al., 2002a; Cowx and Gerdeaux, 2004). Recent trends in the management of recreational fisheries address sustainability by adopting the following management principles already discussed for the management of commercial fisheries (Costanza and Patten, 1995; Caddy, 1999; Charles, 2001; Arlinghaus et al., 2002a; Pikitch et al., 2004).

Precaution and Risk Assessment. The exploitation of irregular or fluctuating resources is largely subjected to uncertainties (Hilborn and Walters, 1992; Ludwig et al., 1993) as are the dynamics of complex ecological and social systems (Holling, 2001). A precautionary approach based on scientific research and analysis should explicitly consider the risks of long term, largely irreversible consequences of management actions and an undesirable and unacceptable outcome of fishing activities (FAO, 1996; Myers and Merz, 1998). A realistic risk assessment should precede any activity that potentially threatens the biological diversity at the genetic, species, or ecosystem level and the burden of proof should shift to those whose activity potentially damages natural resources (Charles 1998; Dayton, 1998). On the other hand, while considering the potential risks to the resource, the expected benefits of the activities should also be taken into account (compare FAO, 1995, 1996). A management under the precautionary principle, which is more stringent than the precautionary approach, should only be used when the uncertainty about the possible outcome is very high and the costs of error may produce irreversible damage (Auster, 2001).

Active Adaptive Management Practice. Many recreational fisheries are managed passively, without specific management plans (Pereira and Hansen, 2003). However, to address the above-mentioned uncertainties about resource dynamics, an active adaptive management practice is required which measures and interprets ecological and social feedback information to modify its activities (Walters, 1986; Bundy, 1998; Walters, 1998). In general, management actions should be based on a periodic assessment of the status of fish populations (Post et al., 2002) and environment (Mace, 2001) and on achievable social and ecological objectives (Lester et al., 2003; Pereira and Hansen, 2003). Clearly, specified management objectives and an evaluation whether current management actions meet the objectives may improve feedbacks between people and the resource basis, and ensure the compliance of anglers to long-term gains in recreational fishing (Hanna, 2001).

Integration of Human Dimensions in Fisheries Management, Interdisciplinarity and Cooperation. Humans are a crucial part of ecosystem dynamics and simultaneously depend on ecosystem services (Holmlund and Hammer, 1999) for their economical and societal development. Consequently, the human dimensions are an integral component of recreational fisheries management (Ditton, 1996; Arlinghaus et al., 2002a; Arlinghaus, 2005). The tight linkage between environment and people and the interconnectedness of aquatic ecosystems on scales that transcend traditional fisheries management boundaries necessitates interdisciplinary approaches in fisheries management (Charles, 1998; Pauly et al., 1998b). In many cases, the factors affecting fish communities and aquatic ecosystems lie outside the control of fisheries management systems and the fishery management authorities usually do not have the political and financial power to implement, per example, restoration programs on their own (Knudsen and MacDonald, 2000). A sufficient communication with and participation of all or the majority of stakeholders can prevent or minimize inter- and intrasectoral conflicts (Dekker and Krueger, 1999; Cowx, 2000; Arlinghaus, 2005). For example, angling as well as nature conservation may benefit from large-scale habitat or ecosystem rehabilitation projects in regulated rivers or food web manipulations in eutrophic lakes. In addition, the involvement of anglers in management decisions may minimize the problem of noncompliance (Cox et al., 2003), which can seriously affect fish populations (Sullivan, 2002). Finally, fisheries management per definition is concerned with people and their goals. Therefore, social needs and perceptions drive fisheries management and the acceptability of actions.

Protection of Large Fish and Genetic Diversity . An increase of the proportion of older repeat spawners and the protection of the natural variability of the size and age structure can help to reduce the probability of recruitment failure when the spawning stock is at low levels (Noble and Jones, 1999; Birkeland and Dayton, 2005). Froese (2004) proposed three indicators for an effective assessment of status and trends in commercial fisheries:

1. to ensure that all fish can spawn before they are caught the catch should consist of only mature fish,
2. only fish of the optimum length (at which a maximum yield can be obtained) should be caught,
3. a percentage of 30–40% of old and large fish (“megaspawners”), which act as a natural safeguard against subsequent recruitment failure, should be protected. The commonly used minimum length limits in recreational fisheries management aim to protect fish until they have reached maturity. Whereas the second point mentioned by Froese (2004) is important only for commercial fisheries, the last point should gain more relevance as a target in recreational fisheries management.

Genetic effects are long-lasting and hardly reversible even when exploitation stops (Conover et al., 2005; Heino and Godø, 2002; Olsen et al., 2004; Reznick and Ghalambor, 2005). The importance of genetics in fisheries management has often been discussed (e.g., Nelson and Soulé, 1987). However, genetic considerations have not always been accorded enough priority in the practice of fisheries management. This may partly be attributed to the fact that much of the evidence of evolutionary effects induced by angling is vague. Selective forces may affect different species and age groups in different ways and vary with exploitation intensity, place, and season of capture, and the biotic and abiotic conditions (Miller and Kapucinski, 1994; Rochet et al., 2000; Reznick and Ghalambor, 2005). However, the principle of the precautionary approach demands taking the risk of genetic changes under consideration (Stokes and Law, 2000), and an "evolutionary enlightened management" (Ashley et al., 2003) should focus on the protection of genetic variability and the prevention of detrimental genetic changes. Until now, there are hardly any generally applicable management actions that avoid the unintentional selection. For example, the harvest of only mature individuals may theoretically select for delayed maturation (Heino, 1998; Ernande et al., 2004; Heino and Godø, 2002). Conover and Munch (2002) suggested that the harvest of the largest individuals should be avoided in order to prevent the harvest-induced decay in somatic growth. This contradiction highlights that minimizing the selection on one trait can increase the selection on another trait (Ernande et al., 2004). To predict evolutionary responses in any fishery demands detailed background knowledge of the stock specific details of the life history strategy, the reaction norms, genetic variances, and covariances of the life history traits, and the selection gradient imposed by the fishery (Munch et al., 2005). In general, the exploitation should not be overly be biased towards a particular component of a population. A fishing pressure that decreases with the biomass may help to decrease the selection pressure (Ernande et al., 2004) and to maintain the local adaptation and a sufficient genetic diversity (Hedrick and Miller, 1992; Reed and Frankham, 2003). In fish populations that show symptoms of high selective exploitation, stricter regulations that limit angler effort and harvest to reduce the overall fishing mortality may be neccessary (Post et al., 2002; Carpenter and Brock, 2004; Hall and Mainprize, 2004). In particular, small or failing populations should receive a greater protection in order to maintain their productive capacity and to conserve adaptive diversity across the geographic range of the species. Problems associated with small population size may persist even if the original cause of population decline is removed (Frankham, 2003). In this regard, not only a single population but also the spatial scales of population structures and the natural network of genetic connections between populations on the basis of recent and historical divergent ecological and genetic data (Crandall et al. (2000) for details) should be taken into account (Youngson et al., 2003).

Ecosystem-Based Management. Ecosystems can shift to contrasting alternative states without early warning signals. A regime shift from a desired to a less desired state can lead to a high cost to society (Scheffer et al., 2001; Scheffer and Carpenter, 2003). In particular the loss of whole functional guilds or species at top trophic levels may strongly affect the function and resilience of ecosystems because slow-changing factors, in particular, sustain ecosystem stability (Folke et al., 2004). Although the interrelation between resilience and biodiversity is still a matter of debate (e.g., Srivastava and Vellend, 2005), there is significant evidence for a positive correlation between biodiversity defined as the variety of life and its processes at the levels of genetic, species, and ecosystems and resilience (Winter and Hughes, 1997; Petchey et al., 2004; Cury et al., 2005; Hooper et al., 2005; Kiessling, 2005; Reusch et al., 2005) as a prerequisite for ecological sustainability that defines the extent to which an ecosystem can absorb recurrent anthropogenic or natural disturbances and adhere to its regenerative efficiency (Hughes et al., 2005). Consequently, recent management approaches point to the importance of primarily managing the resilience of the ecosystems (Callicot and Mumford, 1997; Christensen et al., 1996; Hanna, 1999). Such management approaches are certainly difficult to establish, but may be the best way to protect and conserve fish communities and the social benefits they provide in the long term. As long as for those management approaches mainly abstract guidelines exist, perhaps one of the most fundamental steps may be to overcome concepts focusing primarily on the physical "output" from the fishery such as the maximum sustainable yield (MSY) (Charles, 1998). The MSY should not be viewed as a target to obtain but as a threshold that should not be exceeded and should trigger management actions if reached (Mace, 2001). Although developed for the commercial fisheries, the same principle can be applied to the management of the recreational fishery (Lester et al., 2003). In this context biological reference points as limits or targets (Aprahamian et al., 2006) as well as biological performance indicators for evaluating the state of exploited fish populations may be useful (Gangl and Pereira, 2003). Another step may be to combine aspects of single and multi-species management and include ecological interactions fundamental for the target species (Link, 2002a, 2002b). Without a reliable methodology that can cope with the whole ecosystem, management actions aiming at certain "charismatic umbrella species" (Simberloff, 1998) which fill key functional roles in the system (e.g., key-stone predators, indicator organisms, or integrator species) may improve the conditions for other species and the entire ecosystem (e.g., Freyhof, 2002; Olver, et al., 1995; but see, Roberge and Angelstam (2004)).

Management Strategies

Many strategies have been developed to cope with the increasing angling pressure and to protect or enhance exploited fish stocks. Common strategies focus on three levels: the fishery, the fish stocks, and the aquatic habitats (Cowx, 2002b). The most dominant strategies in recreational fisheries are regulations to limit the number or size of fish harvested by individual anglers and stocking practices (Cowx, 1998; 2002a; Morison, 2004). Habitat management techniques to improve degraded habitats are also used by fisheries managers, although to a lesser extent (Welcomme, 2001).

Bag and Size Limits. Bag or creel limits aiming to regulate the harvest of individual anglers per fishing event or angling day are widely used (Radomski et al., 2001; Radomski, 2003) and can successfully limit the angler effort (Beard et al., 2003b). However, bag limits may not be sufficient enough to limit total harvest (Cox et al., 2002) which may be related to the fact that they may restrict the harvest by the individual anglers, but often do not restrict neither the amount of anglers nor the total harvest (Beverton, 1998; Post et al., 2003; Radomski, 2003). In addition, bag limits may affect only the catch of the experienced anglers, because many anglers do not catch their bag limits (Cook et al., 2001; Radomski et al., 2001, Cox and Walters, 2002). High bag limits may increase the attractiveness of a lake to the anglers and may set a target to attain which can raise unrealistic expectations concerning the catch (Cook et al., 2001). Because the angling satisfaction is linked to the catch and influences the management preferences of anglers, the dissatisfaction following unrealistic expectations may reduce the acceptance of sustainable management practices such as habitat management (Arlinghaus and Mehner, 2005). If the angler effort varies with the bag limits independent of quality or density of the exploited fish population, high bag limits may fail to protect the fish population. In case the bag limits are higher than the biological capability of the fisheries, lowering the bag limits is often suggested to prevent over-exploitation (Cook et al., 2001). However, although a lowering of bag limits may be meaningful from a biological point of view and may work educationally altering the perceptions on fishing success and reminding anglers that their resource is not unlimited (Cook et al., 2001), some aspects may counteract the effects of lower bag limits. Low bag limits may lead to the replacement of small fish with larger fish after the bag limit is reached

(Noble and Jones, 1999). Furthermore, low bag limits may increase angler satisfactions but an increased success may result in a successful angler making more angling trips and attracting additional anglers (Radomski et al., 2001).

Bag limits are often used in combination with size limits (minimum or maximum size limits, slot length limits) which aim to directly reduce the vulnerability of certain size or age groups to angling. Minimum length limits are widely used to reduce angling, total mortality, and the exploitation of fish before they have reached sexual maturity (Noble and Jones, 1999). Results of studies on the effects of length limits or combinations of size and bag limits are inconsistent. Depending on the target species, minimum length limits successfully protected fish populations from overexploitation, improved their size structure, increased the size of harvested fish, the population biomass, and the egg production (Schneider, 1978; Webb and Ott, 1991; Muoneke, 1994; Munger and Kraai, 1997; Cornelius and Margenau, 1999; Miranda and Allen, 2000; Fayram et al., 2001). Other studies reported the failure of size limit regulations (Reed and Davis, 1991; Hale et al., 1999; Margenau and AveLallemant, 2000; Radomski et al., 2001; Bister et al., 2002).

Size or bag limits are most effective if the resulting higher fish densities do not increase the natural mortality and the fish population maintains a sufficient growth. Munger and Kraai (1997) reported that size limits increased the density of legally sized fish, and reduced harvest and catch rates of larger walleyes, but also led to a slight decline in growth. Size limits may fail if the prey availability is limited (Bister et al., 2002). To avoid a growth reduction following a high intraspecific competition the harvest of some fish below the minimum size limits may be recommendable (Stone and Lott, 2002). Some authors assumed that bag and size limits may be appropriate for fish populations characterized by low-to-moderate densities, moderate-to-fast growth, high exploitation, and low natural mortality (Allen and Miranda, 1995; Schultz and Robinson, 2002; Bister et al., 2002). Size limits should not be applied to populations with high densities and slow growth because they may lead to overpopulation and intraspecific competition (Allen and Miranda, 1995; Bister et al., 2002). Size limits may even increase the angling pressure if they increase the attractiveness of the fishery, and anglers interpret larger length limits as an opportunity to catch larger fish and increase their angling effort (Cornelius and Margenau, 1999; Beard et al., 2003a). They may fail in high effort fisheries if catch-and-release fish die as a result of the hooking mortality in significant amounts (Post et al., 2003; Radomski, 2003). Provided there are high harvest rates, size limits may contribute to a rapid removal of fish larger than the size limit (Reed and Rabeni (1989) and literature therein) and therefore fail to protect large individuals potentially favoring size selectivity and corresponding genetic changes.

Inverse slot limits protect fish in an intermediate size class. They may be successful in less productive cold water fisheries and are widely applied for example to centrarchid fish. Their effectiveness depends on good natural reproduction and a favorable growth in the protected size class (Mosindy et al. (1987) and literature therein). Maximum size limits or inverse slot limits may offer some advantages to protect large individuals and to avoid evolutionary changes by intensive angling. The protection of large individuals can broaden the age structure and therefore increase the reproductive potential (Conover and Munch, 2002; Noble and Jones, 1999). Both may favor the fast-growing fish that pass more quickly through the period of vulnerability.

Catch-and-Release. Voluntary or obligatory C&R fishing is widely used for the preservation of exploited fish species (Clark, 1983; Anderson and Nehring, 1984; Jones, 1987; Quinn, 1989; Schneider and Lockwood, 2002; Näslund et al., 2005). Catch and release can reduce the mortality of large fish, improve the size structure (Jones, 1987; Muoneke, 1992) and, from an angler's point of view, increase the total catch and the catch of trophy fish. However, a C&R management is only sufficient if the fish are released unharmed and the detrimental impacts are minimized. The mortality and magnitude of physiological disturbance and the time required for recovery positively correlated with the water temperature (Neal and Lopez-Clayton, 2001; Cooke et al., 2002), duration of catch, fighting, handling, and air exposure (Schreer et al., 2001; Davis and Parker, 2004; Edwards et al., 2004; Schreer et al., 2005). The additive nature of stressors emphasizes the importance of using care, when fish are exposed to angling. Cooke and Suski (2004) summarized some general conditions for an efficient C&R management: fishing methods and equipment that reduce the probability of deep hooking (e.g., artificial lures/flies, barbless hooks, circle hooks), minimized stress, minimized duration of drill, handling, and air exposure. In some cases, cutting the line of deep hooked fish increased the probability of survival (Bartholomew and Bohnsack, 2005).

Catch and release may be less successfull if used during periods of extreme water temperatures, and during the reproductive period. Because sustained swimming may shorten the recovery phase and reduce post-release mortality (Farrell et al., 2001; Graeb et al., 2005; Milligan et al., 2000), a stay in keep nets or recovery boxes before the release can facilitate recovery (Buchanan et al., 2002), provided that material and the dimension of the retention equipment, water quality, and wave action are sufficient (Pottinger, 1997, 1998; Raat et al., 1997; Suski et al., 2005). In general, partial or total C&R management may be a valuable management tool if used properly in combination with the right species, seasons, environmental factors, and sufficiently educated anglers (Anderson et al., 1998; Edwards et al., 2004; Lindsay et al., 2004). However, the species specificity of the fishing effects require adequate data for each species before any mandatory change in the use of tackle or fishing methods to minimize damage can be recommended.

Regulations Targeting Fishing Effort

Under conditions of unsustainable high angling effort and harvest, regulations targeting the fishing effort may be more successful in protecting the fish stocks. Lake-specific harvest quota may be a solution. After the quota is harvested, the lake is closed for fishing (Radomski, 2003). However, in mixed fisheries, the incidental bycatch of protected species may control the fishery and may result in low harvest of other species (Sullivan, 2003). Other solutions are regulations of bait use, lure sizes, total angling effort, or total harvest (e.g., limited licences depending on catch rates), access restrictions, increase in access cost in terms of time or money, lottery systems of access, annual rotation access schemes, or combinations of these options (Cox and Walters, 2002; Lester et al., 2003; Sullivan, 2003; Wilde et al., 2003; Carpenter and Brock, 2004). Those measures may effectively control fishing effort, may be comparatively easier to enforce, and may lead to rebuilding of depleted fish stocks in the long term. However, managers are widely criticized if effort limits are proposed because those management approaches are not in accordance with open access fisheries. Overall, effort controls should only then be favored if catch-and-harvest regulations fail, because of the social implications associated with effort limits.

Protected Areas. One particular form of effort control is the establishment of protected seasons or areas. Seasonal no fishing zones on lakes and streams protect concentrated fish populations during critical time periods (e.g., the spawning season) still allowing angling at other times of the year.

Protected areas such as permanent no fishing zones are less usual in the freshwater fisheries management, although they are becoming increasingly important in the management of marine fish species (Radomski, 2003). Many comparisons between reserves and fished areas were concerned with commercial fisheries; however, some reserves served to protect fish populations depleted by recreational fishing (Jouvenel and Pollard, 2001; Westera et al., 2003). Compared with adjacent areas, marine reserves lead to rapidly developing and persistent effects such as average increases in densities, biomass, individual size, and species diversity in all functional groups (Côté et al., 2001; Halpern, 2003) and reduce collateral ecological impacts such as habitat destruction by fishery related activities (Hilborn et al., 2004). The abundance and age structure of exploited and large-bodied species respond especially positively to reserves (Evans and Russ, 2004; Mosqueira et al., 2000). The reserves can form an insurance against the detrimental effects of selective fishing and contribute to the protection of the genetic diversity, if their design encourages and sustains large populations, if there is no extremely strong density dependence in juvenile or adult fecundity, and if there is a sufficient gene flow over the boundaries in the adjacent area (Conover and Much, 2002; Gell and Roberts, 2003; Roberts et al., 2005; Pérez-Ruzafa et al., 2006). In addition, reserves influence the adjacent areas via the spillover of eggs, larvae, and fish from the reserve to adjacent areas outside the reserve boundaries. The spillover can directly or indirectly enhance the yield of local fishery and the recruitment of fish populations outside the reserve (Russ et al., 2004; Ashworth and Ormond, 2005). However, reserves may vary in their effectiveness, depending on species, functional guild, movement, home range and type, size, variety, and number of habitats included in the reserve (Francour et al., 2001). They favor short-lived and fast-growing species with a comparable small home range while the protection of larger and more mobile fishes may afford other policy options (Halpern and Warner, 2002).

As far as the establishment of inland freshwater reserves is concerned, the protection of fish species has received less attention relative to other vertebrates (Maitland, 1995) and, to our knowledge, there are very few scientific studies concerning the effect of protected areas on freshwater fish populations (Keith, 2000; Crivelli, 2002). In general, the protection of certain areas of rivers or lakes may contribute to the rebuilding of fish stocks, especially for fish species which are affected by fishing, habitat loss, and other threats to aquatic biodiversity (Suski and Cooke (2006) and literature therein). However, existing reserves are ineffective if they protect only a fraction of the home range which is of critical importance for the target species. Reserves are only effective if they have the size and biological diversity to protect a sufficient part of the population and contain a range of habitats essential for the whole life cycle, improve migration corridors, increase the recolonization of adjacent habitats, and establish refuge populations (Aparicio et al., 2000). In particular isolated reserves should be capable of supporting a sufficient number of adult animals (Reed et al., 2003; Reed and Hobbs, 2004). However, if exchange rates between reserves and open areas are large, the effect of protected areas may be lower than expected (Cox and Walters, 2002). Consequently, protected areas must be complemented by parallel policies taken in areas outside the reserves to ensure the conservation of ecosystems and fish communities in the long term (Margules and Pressey, 2000).

Enhancement Practices: Stocking and Habitat Restoration

Stocking and introduction of fish are widespread management tools in recreational fisheries (Cowx, 1998) and can mitigate the effects of recruitment failures, disturbances to fish populations or water bodies caused by anthropogenic activities, can improve the quality of the fishery, and enhance threatened fish populations (Welcomme and Bartley, 1998). There are numerous guidelines available that help to mitigate ecological and genetic risks associated with stocking (e.g., Meffe, 1986; Wright and Giles, 1987; Cowx, 1994; Moring et al., 1995; Cowx, 1999; European Inland Fisheries Advisory Commission, 1994; Wickström, 2001; Molony et al., 2003; Aprahamian et al., 2004; Connor et al., 2004; Mehner et al., 2004). The genetic aspects are especially of crucial importance. The disruption of local adaptation can be avoided by the use of local brood stocks. Mating schemes and breeding conditions can foster the local retention of adaptive genetic variation. More natural conditions in the hatchery environment can reduce domestication effects and increase post-release survival by facilitating the training of life skills such as foraging or predator avoidance (Brown and Laland, 2001; Brown et al., 2003a; 2003b; Wiley et al., 1993). In accordance with the precautionary approach, stocking should be accompanied by risk assessments and appropriate monitoring programs and unjustified introductions or transfers among populations should be avoided.

In some cases much of the budgets of fisheries managers are spent on stocking (Epifanio, 2000). However, stocking has often been considered unsucessful due to low habitat quality, increased intra- or inter-specific competition, prey switching of natural predators, food or habitat limitation due to overstocking, handling mortality, poor adaptation to natural conditions of the captively reared fish, and genetic bottlenecks (Moyle et al., 1986; White and Schell, 1995; Armstrong and Knapp, 2004; Van Zyll de Jong et al., 2004). In addition, the short term success following stocking may result in increasing effort and exploitation that contradicts the management success. In this context, a shift from a predominance of artificial propagation, rearing, and stocking (Williams et al., 1999) to the rehabilitation of aquatic habitats may be promising. The rehabilitation of aquatic habitats which aims at the ecological integrity of aquatic ecosystems and encompasses the increase of habitat diversity and the improvement of water quality (Roni et al., 2002) may be more successful than the traditional stocking practices while the ecological risks are minimal (Arlinghaus et al., 2002a; Cowx, 1994).

The restoration of aquatic ecosystems to pristine conditions is a utopian view (Cowx and Van Zyll de Jong, 2004). However, there are many examples of successful habitat improvements in running and standing waters (Jeppesen et al., 1999; Jurvelius and Auvinen, 2001; Kairesalo et al., 1999; O'Grady and Duff, 2000; Souchon and Keith, 2001; Roni et al., 2005). Habitat improvements used to increase the production potential of water bodies range from the protection or creation of spawning substrates or littoral habitats that are a critical but vanishing habitat element in both lakes and streams (Christensen et al., 1996) to complex tasks such as the restitution of natural flow regimes and connectivity in regulated rivers, and the reduction of external and internal nutrient loads in lakes and reservoirs (Cowx and Welcomme, 1998). Although the rehabilitation of all degraded rivers or artificial waterways may be presumably unrealistic, a relatively simple instream habitat improvement which modified flow and bottom substrate or provided cover had long-term beneficial effects on fish communities (Swales, 1989; Roni et al., 2002; Pretty et al., 2003). For example, a rehabilitation of 20% of the bank line in artificial waterways may lead to a substantial improvement of diversity and persistence of a fish community (Wolter, 2001). The enrichment of structures can contribute to the enhancement of fish populations and to an increase of angling quality not only in running but also in standing waters (Bolding et al., 2004). Standing waters are to a great extent affected by human-induced nutrient input (Mehner and Benndorf, 1995) which influences lake productivity and fish communities (Persson et al., 1991). Because a reversal of cultural eutrophication may lead to a decrease in biomass of certain fish species (e.g., pikeperch), anglers may have different perceptions about the directions of management aims, depending on their main target species. However, changes due to eutrophication are often socially undesirable (e.g., algal blooms), so that anglers may accept that a reversal of eutrophication is of social priority. On the other hand, among the different measures used for the restoration of eutrophicated lakes, a top-down food web management (biomanipulation) (Lammens, 1999) combined with the reduction of external nutrient loads which influences food web structure and nutrient cycles via increase of the density of piscivorous fish species may be beneficial for water quality and recreational fishery (Mehner et al., 2004). In general, degraded ecosystems should be rehabilitated if the scientific knowledge is sufficient and the economic and social conditions allow for the rehabilitation projects. The monitoring should be improved to examine the effectiveness of the rehabilitation including an appropriate budget when planning rehabilitation actions. If large-scale habitat rehabilitation schemes are not possible to implent due to biological or socio-economic constraints, management actions such as stocking should be applied provided that there is an appropriate risk assessment (Cowx and Van Zyll de Jong, 2004).

Habitat restoration will probably require more resources or a shift in resource use (Radomski, 2003). Therefore, sophisticated education programs and public outreach are needed to convince stakeholder groups to shift the allocation of financial resources (Holland, 1996). However, habitat improvement may significantly improve not only the fishery but also benefit other species (Noble et al., 2004) and may have wider social and economic benefits which will help to increase the social acceptance (Pitcher and Pauly, 1998).

Rehabilitation activities may not fully compensate a former loss of ecological functions. Fish communities may react differently with an increase of biomass at the expense of biodiversity, or with an increase of species complexity, to habitat rehabilitation (Cowx, 2002b). Therefore, management actions should be primarily directed towards a protection of the relative pristine ecosystems. The prevention of losses and damages is much more preferable and cost-efficient compared to rehabilitation activities. The protection of slowly changing variables such as coarse woody debris especially creates ecological resilience and thereby provides managers with a broader range of options (Carpenter and Brock, 2004).

Adaptation of Recreational Fisheries Management to Local Conditions: Illusion or Future?

Recreational fisheries management practices are usually based on a one-size-fits-all policy. However, fish populations of different water bodies vary locally with environmental, biological and fishery related conditions and so does the reaction of fish populations to fisheries regulations (e.g., Goeman et al., 1993; Allen and Miranda, 1995; Wilde, 1997; Cook et al., 2001; Bartholomew and Bohnsack, 2005). Consequently, management measures should be adjusted to individual populations and to the productivity of the water body to assure an adequate protection (Paukert et al., 2001; Beard et al., 2003a, 2003b). On the other hand, the recreational fishery and its impact on a fish population does not function in isolation. Because of the rapid information transmission through the angler community (Carpenter et al., 1994) a decreasing angling quality may shift the angling effort to nearby water bodies (Post et al., 2002) which ultimately may cause a spreading of collapses about various water bodies in a region if local regulations are insufficiently stringent. Too stringent regulations on the other hand may enhance illegal harvest and may fail in preventing the landscape-wide degradation of fisheries (Carpenter and Brock, 2004). Faced with this dilemma, a flexible management, which is temporally and spatially matched to the scales of ongoing exploitation and allows an adaptation to changes in social and ecological conditions, may be a solution (Carpenter and Brock, 2004). The high complexity in regulations may be unacceptable to anglers (Lester et al., 2003); therefore education and public participation are neccessary to allow difficult solutions and strong restrictions to be more easily accepted (Arlinghaus et al., 2002a; Radomski, 2003).

Conclusions and Future Research Needs. Although the importance of angling is increasing worldwide, angling has received less academic attention than commercial fishing, in part because recreational fisheries have been viewed as self-sustaining (Post et al., 2003). The main threats to fish populations, particularly in freshwater ecosystems, originate from outside the recreational fisheries. However, based on this review and others (Cooke and Cowx, 2006), there are some reasons to suggest, that recreational and commercial fishing may have some serious impacts on fish populations and ecosystems in common, provided that the degree of the fishing mortality is similar, which is often, but not always, the case (Lester et al., 2003). Table 1 provides a summary of the issues identified and discussed in the present article together with qualitative indicators of the severity and generality of the impacts discussed. It also points to those impacts that require more research as well to the occurrence and importance of the effects in relation to recreational fishing on a global scale. The impacts are classified as severe if they are long-lasting and characterized by a low reversibility. These are primarily impacts associated with the risks of genetic changes and the loss of biodiversity. High and selective exploitation rates can be followed by depensatory effects which may lead to a loss of populations and may prevent a rebuilding even after a cease of the exploitation. In combination with a high selectivity, angling can change the age structure of fish communities, which subsequently may contribute to a loss of reproductive capacity, and the composition of fish communities, which may result in an alteration of the aquatic food web and a loss of biodiversity. In addition, there is a risk of detrimental changes in some life history characters, some of which might be transmitted by genetic changes to future generations. Even if high percentages of caught fish are released, immediate and delayed post-release mortality or sublethal effects may affect the populations. While the consequences of high exploitation rates and selectivity are much more serious and require more sophisticated management approaches, the impacts resulting from the angling activity per se may be primarily of local importance and comparably easily to manage. Stocking and the transfer of live bait fish are classified as severe, because of their potential impacts on the genetic integrity of native populations.

However, the occurrnce of some biological impacts of angling are not well investigated. While there are many studies on age and size truncation and C&R fishing, the potential impacts of angling on habitats, trophic cascades, and on the genetic structure of exploited population, especially the questions of evolutionary changes, the implications of fishing down aquatic food webs (Pauly et al., 1998a) and the responses of ecosystem resilience to a potential loss of species diversity need further exploration. In addition, we found a strong geographical bias towards North America in the research on both potential impacts and management of recreational fishing. Therefore, we advocate similar research in other regions, for example, in Europe, as well. On a worldwide scale, information on, inter alia, the status of exploited fish stocks, the level of angling exploitation and harvest, and ecological and evolutionary response of fish and ecosystems to angling and management actions is often lacking.

Although the common management approaches used in recreational fisheries were often successful, they could not always prevent overexploitation (Lester et al., 2003). In general, more active and adaptive management approaches are needed to avoid population collapses and to maintain the resilience of aquatic ecosystems, healthy fish stocks, and high quality fishing for long term. Those adaptive management approaches are useful in enlightening the reaction of aquatic ecosystems to human interventions. The importance of primarily managing the resilience of the ecosystems is widely accepted (Hanna, 1999; Pikitch et al., 2004). However, while there have been many models and measures developed for the commercial fisheries (e.g., Mace 2001; Hall and Mainprize, 2004; Stefansson and

Table 1

Documented impacts and potential risks resulting from angling and angling-related activities together with an indication of global importance

Pattern	Documented effects	Potential risks	Importance	References
High exploitation	Decline in catch and harvest, high population fluctuations in some target species	Depensation, population collapse, loss of genetic diversity	***	Allen et al., 1998; Beard Jr. and Kampa, 1999; Gunn and Sein, 2000; Post et al., 2002; Radomski, 2003
Selectivity for size and size-related life history traits	1. Shifts in length and age distribution in some target species 2. Decrease in age at first maturation, alteration of the genetic growth potential, and some related reproductive traits	1. Truncation of the natural age structure, 2. Detrimental changes in size related life history characters, loss of evolutionary potential	*** not well investigated in freshwater, more research needed	1. Anderson and Nehring, 1984; Goedde and Cobble, 1981; Mosindy et al., 1987; Olson and Cunningham, 1989; Pierce et al., 1995; Sullivan, 2003; Almodovar et al., 2004 2. Diana, 1983; Nuhfer and Alexander, 1994 ; Drake et al., 1997 ; Magnan et al., 2005
Selectivity for behavioral traits	Selection pressure against boldness, aggression, changes in migration time	Detrimental changes of the natural behavioral traits	***	Mezzera and Largiadér, 2001; Garrett, 2002; Consuegra et al., 2005
Selectivity for sex	Higher vulnerability of either males or females to angling	Decrease in the effective population size, effects on recruitment, loss of genetic diversity	**	Casselman, 1975; Serns and Kempinger, 1981; McCleave and Jellyman, 2004; Peréz et al., 2005
Selectivity for species	Selective removal of species	Changes in species composition and trophic cascades, loss of ecosystem resilience	*** not well investigated in freshwater, more research needed	Pinnegar et al., 2000;

Bycatch, catch-and-release fishing	Stress and injuries in target and non target species,	Immediate or delayed mortality, sublethal fitness impacts	**	Muoneke and Childress, 1994; Cooke et al., 2003c; Edwards Jr. et al., 2004; Lindsay et al., 2004; Steinhart et al., 2004; Bartholomew and Bohnsack, 2005
Disturbance of habitats and wildlife	Damage to macrophytes or woody structure, disturbance of waterbirds and other wildlife	Habitat loss, effects of nutrient cycling, stress and recruitment failure in waterbirds and other taxa	*	Rees and Tivy, 1978, Liddl and Scorgie, 1980; Sukopp and Markstein, 1981; Brosnan and Crumrine, 1994; Sudmann et al., 1996; Müller et al., 2003
Boat traffic	Wave action, emission of inorganic and organic compounds, dewatering of fish eggs and larvae	Negative impacts on littoral vegetation and sediment, recruitment failure and fitness loss of some species, decrease of water quality	*	Mueller, 1980; Yousef, 1980; Tjärnlund et al., 1995; Wolter and Arlinghaus, 2003
Noise (associated with boat traffic)	Stress, loss of hearing abilities	Negative effects on survival and fitness	*	Scholik and Yan, 2002a; 2002c; Smith et al., 2004a; 2004b; Wysocki et al., 2006
Loss of fishing gear (lines and lead sinkers)	Damage to wildlife, lead emission	Negative effects on water quality, survival and fitness of piscivorous birds, danger to humans	*	Cryer et al., 1987a; Sears, 1988; Mowad et al., 1998; Scheuhammer et al., 2003; Asoh et al., 2004; Chiappone et al., 2005
Bait digging	Disturbance of littoral fauna and sediment	Negative effects on population dynamics, survival and fitness of invertebrates	*	Cryer et al., 1987b; Keough et al., 1993; Beukema et al., 2002; Contessa and Bird, 2004

(Continued on next page)

Table 1

Documented impacts and potential risks resulting from angling and angling-related activities together with an indication of global importance *(Continued)*

Pattern	Documented effects	Potential risks	Importance	References
Nutrient input	Input of nutrients, negative impacts on benthic invertebrates and water quality	Eutrophication, decrease of water quality, ecosystem changes	*	Cryer and Edwards, 1987; Arlinghaus and Mehner, 2003; Niesar et al., 2004
1. Stocking 2. Transfer of bait fish	Introduction of nonindigenous species, negative impacts on local fish stocks due to competition, predation or hybridization, spread of diseases	Decrease or loss of native species following hybridization, spread of parasites and diseases, changes in fish community and ecosystem, changes in fish community and ecosystem	***	1. Barthelmes et al., 1998; McNaught et al., 1999; Novinger and Rahel, 2003; Simon and Townsend, 2003; Weber and Fausch, 2003; Gilk et al., 2004; Gozlan et al., 2005 2. Ludwig and Leitch, 1996; Winfield and Durie, 2004, Goodwin et al., 2004; Holthe et al., 2005

Asterisks indicate the ecological importance according to the subjective opinion of the authors.

***severe threat to fish communities and ecosystems due to low reversibility, sophisticated management approaches required.

**medium severe effects

*effects of primarily local importance, reversible and easier to manage.

Rosenberg, 2005), analogous performance measures for the recreational fisheries for which currently mainly abstract principles and guidelines exist are lacking (Arlinghaus et al., 2002a; Welcomme, 2001; but see Gangl and Pereira, 2003; Radomski, 2003). Consequently, many regulations are often set more or less arbitrarily (Radomski et al., 2001). There is a need to develop quantifiable ecosystem measures that not only characterize single species dynamics but also species interactions and nontarget species (Murawski, 2000), account for ecosystem resilience (Hughes et al., 2005), trigger particular management actions when reached, and that can be used to judge the success of given management strategie (Caddy, 2002) and as early warning signals of ecological problems (Niemi and McDonald, 2004).

Fisheries management is seen to be as much about managing people as about managing fish stocks (Larkin, 1977; Ditton, 1996; Arlinghaus et al., 2002a), which points to the importance of integrating psychological, sociological, and economic perspectives in recreational fisheries management (Cox and Walters, 2002; Post et al., 2002; Arlinghaus, 2005; Cury et al., 2005). To provide a more realistic picture of angling exploitation and to identify promising management strategies, more research is needed to explore the dynamics between anglers and their species and in particular the question how anglers respond to the dynamics of their target species and management actions. In addition, the interactions between catchability, angling effort and fish densities have been quantified for only some water bodies and fish species (Pereira and Hansen (2003) and literature therein). Because anglers are heterogeneous in their interest and respond in complex ways to management actions (Beard et al., 2003a; 2003b), a better understanding of angler heterogeneity and the consideration of angler attitudes toward various management options and potential outcomes into management practices would improve management success. The integration of the social and economic dimensions may contribute to building up alliances with interested parties to limit damage to aquatic ecosystems, promote rehabilitation activities, and to strengthen the feedback among managers, stakeholders, and resource basis. Research in the field of communication and information may elucidate angler preferences and their responses to management actions and may be helpful for a successful design of educational campaigns to establish a system of incentives to promote a human behavior consonant with sustainable use of aquatic ecosystems. Often the angler community does not fully recognize the need to protect biodiversity, and considers indigenous fishes of marginal importance (Cambray, 2003). With the development of education campaigns for anglers and local fisheries managers, especially local voluntary managers of small angling clubs, it may be possible to modify their incentives and behavior to minimize threats to fish populations and aquatic ecosystems and to establish a sustainable management.

Even though recent research has highlighted the tight linkage between socio-economic factors, land use practices, aquatic ecosystems, and fish populations (Cowx and Van Zyll de Jong, 2004), the future role of recreational fisheries management in watershed management, rehabilitation, and conservation strategies in the view of future changes in social and economic factors, land use practices, and potential impacts of climatic changes is largely unexplored. It is our hope that this review stimulates an urgently needed discussion about the pros and cons of recreational fishing and about ways to manage for sustainability by harmonizing the interests of recreational fisheries with the “interests” of the natural world.

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