Reconciling traditional inland fisheries management and sustainability in industrialized countries, with emphasis on Europe

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Abstract

In northern industrialized countries, the inland fisheries sector has long been dominated by recreational fisheries, which normally exploit fish for leisure or subsistence and provide many (poorly investigated) benefits to society. Various factors constrain the development and existence of inland fisheries, such as local user conflicts, low social priority and inadequate research and funding. In many cases, however, degradation of the environment and loss of aquatic habitat are the predominant concerns for the sustainability of inland fisheries. The need for concerted effort to prevent and reduce environmental degradation, as well as conservation of freshwater fish and fisheries as renewable common pool resources or entities in their own right is the greatest challenge facing sustainable development of inland waters. In inland fisheries management, the declining quality of the aquatic environment coupled with long-term inadequate and often inappropriate fisheries management has led to an emphasis on enhancement practices, such as stocking, to mitigate anthropogenic stress. However, this is not always the most appropriate management approach. Therefore, there is an urgent need to alter many traditional inland fisheries management practices and systems to focus on sustainable development.

This paper reviews the literature regarding the inputs needed for sustainability of inland fisheries in industrialized countries. To understand better the problems facing sustainable inland fisheries management, the inland fisheries environment, its benefits, negative impacts and constraints, as well as historical management, paradigms, trends and current practices are described. Major philosophical shifts, challenges and promising integrated management approaches are envisaged in a holistic framework. The following are considered key elements for sustainable development of inland fisheries: communication, information dissemination, education, institutional restructuring, marketing outreach, management plans, decision analysis, socioeconomic evaluation and research into the human dimension, in addition to traditional biological and ecological sciences. If these inputs are integrated with traditional fisheries management practices, the prospects for sustainability in the inland fisheries will be enhanced.

Keywords angling, ecosystem management, freshwater fisheries, habitat rehabilitation, recreational fisheries, stocking

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Introduction

Since ancient times, fishing has been a major source of food and income for society (Cowx 2002c). However, its importance relative to other food production systems has waned in the last half century (FAO 1997). This is especially true for fishing activities in inland waters of densely populated and highly industrialized countries of the northern temperate world, where multipurpose use patterns have created a very distinct climate for the development of inland fisheries (FAO 1997). Activities such as agriculture, damming, channalization, deforestation, navigation, wetland reclamation, urbanization, hydropower generation, water abstraction and transfer and waste disposal have altered freshwater ecosystems profoundly, probably more than terrestrial ecosystems (Vitousek *et al.* 1997; Cowx 2000). As a result, the majority of freshwater ecosystems in industrialized countries are considered impacted (Dynesius and Nilsson 1994; Vitousek *et al.* 1997), and (genetic, species and community) biodiversity change between 1990 and 2100 is estimated to be least in northern temperate ecosystems because major land-use changes have already occurred (Sala *et al.* 2000). Therefore, in most areas of the world, the principal impacts on inland fisheries do not originate from the fishery itself but from outside the fishery (e.g. FAO 1997; Garcia *et al.* 1999; Welcomme 2001). For the last 20 years, as a result of the established environmental awareness within industrialized societies (Diekmann and Franzen 1999), the realization of the poor state of most freshwater ecosystems and the implementation of remedial actions (e.g. sewage treatment to reduce nutrient loading), environmental quality has increased in many fresh waters throughout the industrialized world, both in running (e.g. Cowx 2000, 2002a; Raat 2001) and standing waters (e.g. Eckmann and Rösch 1998; Müller and Bia 1998; Jurvelius and Auvinen 2001; Cowx 2002a,b). These developments which were often initiated by nonfishery stakeholders and implemented by nature conservation, environmental or water authorities, were supported by numerous international conventions and directives which aim to protect water quality (e.g. European Water Framework Directive 2000/60/EEC of 22 December 2000), aquatic habitats (e.g. EC Habitats Directive Conservation of Natural Habitats and of Wild Flora and Fauna 92/43/ECC of 21 May 1992, adapted to scientific progress with EU Council Directive 97/62/EC of 27 October 1997) and biodiversity, including freshwater fish (e.g. EU Council Decision 98/746/EC of 21 December 1998 concerning amendments to the Berne Convention on the Conservation of European Wildlife and Natural Habitats, 'Natura 2000' framework, IUCN Red List). Unfortunately, nature conservation programmes and improvement of water quality (e.g. reduction of pollutants, Cowx 2002a) have rarely been successful at substantially protecting and enhancing freshwater fish (e.g. Raat 2001; Souchon and Keith 2001; Cowx and Collares-Pereira 2002). Reasons may include fish species diversity being more dependent on rehabilitation of habitat structure and maintenance of lateral and longitudinal connectivity than on improvement of water quality (e.g. Lucas and Marmulla 2000; Wolter 2001; Collares-Pereira *et al.* 2002a). Furthermore, the number of examples of conservation actions targeting fish in the literature is small in comparison to other animal species, and the number of examples where fish populations have increased and expanded is pitiful (Kirchhofer and Hefti 1995; Cowx 2002a; Cowx and Collares-Pereira 2002). Therefore, despite marginal increases in inland fish catches in many countries and several regions of the world (about 2% per year, Fig. 1), which are the result of fishery enhancements by stocking, human-induced eutrophication and simply better statistical information, degradation of the environment and loss of aquatic habitat still remain the predominant concerns for the sustainability of inland fisheries (FAO 1999). The need for concerted effort to prevent and reduce environmental degradation – as well as conservation of freshwater fish and fisheries as renewable common pool resources or entities in their own right – is the greatest challenge facing sustainable development of inland waters (FAO 1999).

Inland fisheries can be viewed as evolving organisms (Fig. 2), with the major stages in the life cycle of an inland fishery comprising an initial emphasis on food production, then a growing interest in recreation, with aesthetic and nature conservation interests emerging last (Smith 1986). This is also seen when the objectives of inland fisheries management in developed and developing countries are compared (Table 1). Fisheries management in industrialized countries focuses almost exclusively on recreation and conservation, whereas developing countries still focus on food security, although the emphasis on recreational fisheries (Cowx 2002c) and conservation (Collares-Pereira *et al.* 2002a) are increasing as a result of globalization. Basically, as commercial productivity and the number of commercial and recreational users increase, conservation of the resource requires more stringent management intervention. Furthermore, with increasing exploitation pressure on inland fishery resources, both in terms of effort and fishing efficiency, food production opportunities tend to decline and recreational uses expand (Smith 1986; Radonski 1995). Consequently, in most temperate countries recreational, leisure or 'sport' fisheries are the dominant components of evolved

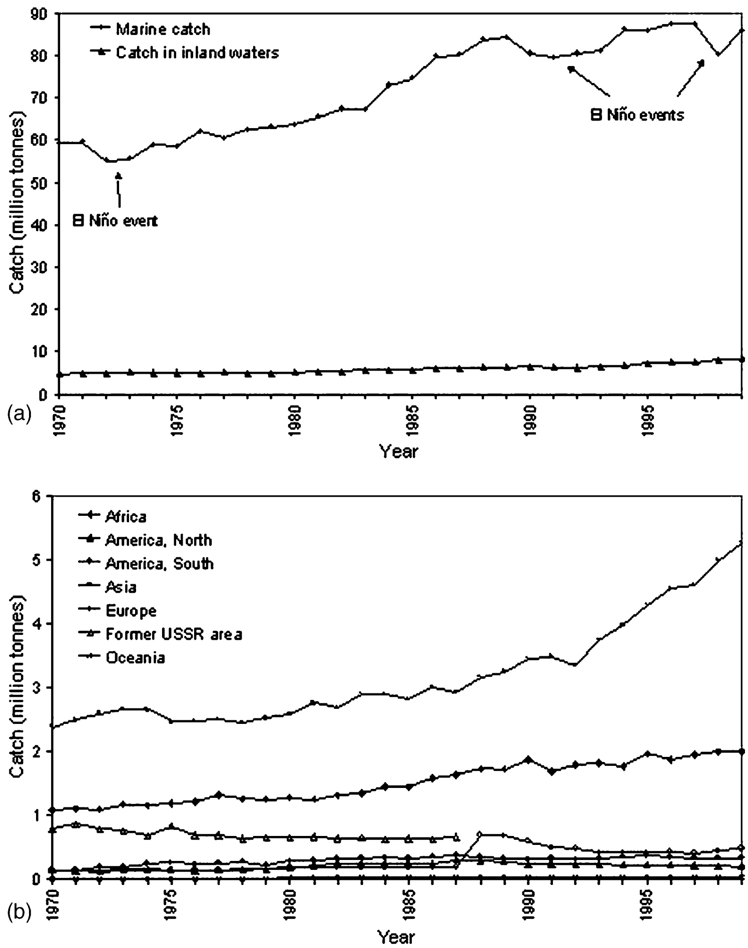

Figure 1 Time-series (1970–99) of marine fisheries catches and catches in inland waters (a) and of catch in inland waters by continental region (b) European data for 1988 onwards include former USSR data) (Source: FAO 2000; El Niño events according to Watson and Pauly 2001). Note that underreporting of catch by countries and incomplete data on recreational fisheries suggest that catches in inland waters may be at least twice as high as shown on the figure (FAO 1999).

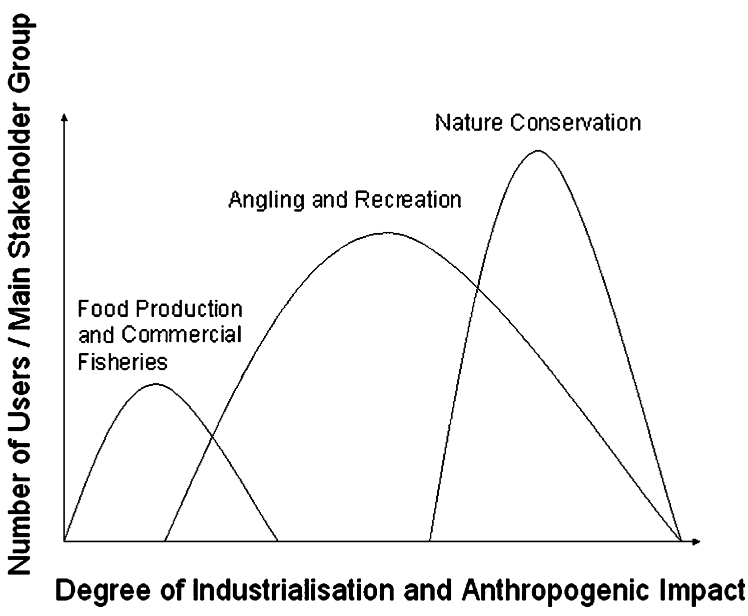

Figure 2 Generalized life cycle of inland fisheries (modified from Smith 1986). Evolution takes place along an industrialization gradient where user numbers increase and stakeholder dominance changes.

Table 1 Different strategies for management of inland waters for fisheries in developed and developing countries (from Welcomme 2000, 2001; slightly modified).

	Developed (temperate)	Developing (tropical)
Objectives	Conservation/Preservation	Provision of food
	Recreation	Income
Mechanisms	Recreational fisheries	(Commercial) Food fisheries
	Habitat rehabilitation	Habitat modification
	Environmentally sound stocking	Enhancement, e.g. through intense stocking
	Intensive aquaculture	Extensive, integrated, rural aquaculture
Economic	Capital intensive	Labour intensive

inland fisheries systems and have long represented the major use of living aquatic resources (FAO 1999; Welcomme 2001; Cowx 2002c).

In some European countries, e.g. Germany, this evolution of inland fisheries has been almost completely neglected. For example, compared with the abundant North American literature and extensive experience regarding recreational fisheries management, efforts to understand and better manage European recreational fisheries seem negligible (Aas and Ditton 1998), despite the high socioeconomic and sociocultural benefits created by angling (see Section 3). Moreover, in Europe, investigations on recreational fisheries have developed slowly (Aas 2002), despite the obvious need for an integrated management approach to promote sustainable inland fisheries management systems, which take into account the multifaceted nature of inland water uses. Irrespective of the high social importance, recreational fisheries have a number of negative ecological impacts (see Section 3). Therefore, sustainable inland fisheries management systems constitute both a challenge and a duty. The need for management of natural freshwater ecosystems and all systems severely impacted by humans reflects an inability of the systems to operate in self-sustaining ways due to interference or damage to an extent that is beyond the capabilities of the system to self-repair (Moss 1999). The challenge and duty of sustainable inland fisheries management cannot be regarded just as an end in itself, but is just a symptom of failure of the fisheries management and ecosystems in general (Moss 1999).

The aims of this paper are: (i) to provide an insight into the interdependence of fisheries and the concept of sustainability to maintain, improve and develop inland fisheries; (ii) to explore opportunities and constraints in achieving sustainable use of inland waters with special reference to recreational fisheries in Europe and (iii) to encourage further scientific investigations into sustainable inland fisheries management.

The paper will: (i) explore the theoretical basis of the sustainability concept with respect to fisheries; (ii) describe developed inland fisheries systems; (iii) describe the evolution of inland fisheries management and its paradigms; (iv) outline current inland fisheries management practices; (v) elucidate possible management directions for freshwater fisheries with special emphasis on traditional fisheries management systems and (vi) recommend prospects and perspectives, as well as research needs. The diverse nature of inland fisheries systems worldwide (O'Grady 1995; FAO 1999) necessitates that to remain meaningful, the paper focuses on one region, Europe. However, abundant North American and Canadian fisheries literature is considered.

Inland fisheries and sustainability

Fisheries systems worldwide are characterized by complex interrelationships between society and the natural environment (Lackey 1979; Caddy 1999; Charles 2000; Cochrane 2000). This complexity, their long history, the high degree of development within the life cycle of fisheries and the poor state of many fisheries systems (e.g. Pauly and Christensen 1995; Garcia and Newton 1997; Buckworth 1998) and freshwater ecosystems in developed countries make them ideal case-studies for the concepts of sustainability and sustainable development (SD) (Charles 1994), both being inherently complex conceptual and normative approaches (Barrett and Odum 2000). However, there is no general consensus for a clear definition of sustainability and there are hundreds of general definitions of SD (see Garcia and Staples 2000) which make both concepts almost convert to 'buzz' words. SD was popularized by the

Brundtland Report (WCED 1987) and subsequently by the United Nations Conference on Environment and Development ('Earth Summit' Agenda 21 in Rio de Janeiro 1992). According to the Brundtland Report, a development is sustainable when it 'meets the needs of the present generation without compromising the ability of future generations to meet their own needs' (WCED 1987). It is important to mention that the definitions of the Brundtland Report and of the Rio Conference are explicitly anthropocentric (Rennings and Wiggering 1997). Anthropocentric definitions of SD include no reference to environmental quality, biological integrity, ecosystem health or biodiversity (Callicot and Mumford 1997). This is also true when looking at weak sustainability (WS) which, together with strong sustainability (SS), constitute two distinct approaches to sustainability (Goodland and Daly 1996; Garcia and Staples 2000; Hediger 2000). Weak sustainability, which is more likely to be embraced by conventional economists (Jamieson 1998), implies that natural, man-made, human and social capitals are perfect substitutes for each other. Weak sustainability, like other anthropocentric definitions of SD, makes no essential reference to environmental goods (Jamieson 1998). Therefore, WS allows radical depletion of natural capital provided the sum of the above four capitals (and the services which could be provided by it) is kept constant for future generations or increases over time (Garcia and Staples 2000). Strong sustainability, on the other hand, is more related to the biocentric viewpoints of ecologists and ecological economists who value the biota and natural capital for its own sake (Costanza and Daly 1992; Goodland and Daly 1996; Callicot and Mumford 1997). Strong sustainability assumes that all forms of capital are not equivalent but complementary and should be conserved in their own right (Costanza and Daly 1992). Pajak (2000) favoured the 'simple and widely used, . . . succinct and understandable' anthropocentric SD definition of WCED (1987) when reconciling sustainability, ecosystem management and fisheries. Garcia and Grainger (1997), on the other hand, stated that the whole theory of fisheries management (e.g. maximum sustainable yield (MSY)) is based on SS concepts and thus on a rather biocentric philosophical viewpoint. Regardless of the different theoretical perceptions on sustainability, both philosophical viewpoints have the power to rebuild, revive and conserve inland fisheries systems on the larger scale and are not counter-intuitive to the future development of fisheries *per se*. This is true because in fisheries systems, on the larger scale, only healthy aquatic ecosystems are able to produce high social and economic benefits (community welfare). Healthy and functional ecosystems are, at the same time, the prerequisite for ecological sustainability (biocentric SD; Callicot and Mumford 1997; see also Goodland and Daly 1996 for environmental sustainability). Conversely, the implementation of an anthropocentric SD means that the interests and well being of fishers now and in the future are taken into account. However, there remains the risk that according to WS concepts 'sustainable' management practices on the local scale lead to unsustainability on the larger scale. Consequently, given the minimum necessary condition, which is the maintenance of the total natural capital stocks (i) at or above the current level (Costanza and Daly 1992) or (ii) at or above critical levels thought to be consistent with ecosystem stability and resilience (Navrud 2001b), every properly interpreted and adopted sustainability concept is an opportunity for the conservation of threatened fisheries systems, including inland fisheries worldwide. Taking this into account, the challenge for inland fisheries is not to debate direction and definition of sustainability or SD, but to finally achieve and implement sustainability by shifting from a 'sectoral' view, in which the fishery is treated in isolation, to an integrated, multidisciplinary 'systems' view (Charles 1998, 2000). This should lead to qualitative (i.e., sustainable) development of inland fisheries, which is fundamentally different to (quantitative) throughput growth that is often meant when speaking about development (see Goodland and Daly 1996 for details).

In the context of sustainability, rebuilding ecosystems should be the overarching goal of modern sustainable fisheries management and not sustainable fisheries *per se*, because public support is more likely to occur for sustainable ecosystems than for sustainable fisheries (Pitcher and Pauly 1998; Pitcher 2000, 2001). Furthermore, the total value of the resource will rise quickly if functioning, diverse aquatic ecosystems, including commercially and recreationally valuable top predators, are restored (Pitcher and Pauly 1998; Pitcher 2001). Therefore, because of the threats to inland fisheries originating mainly from outside the sector, sustainable inland fisheries management systems have to be considered as integrated part of a holistic management of (specific) aquatic ecosystems or watersheds (Fig. 3, compare, e.g. Caddy 1999; Garcia *et al.* 1999; Pitcher 2001; Scheffer *et al.* 2001), which are encapsulated in three domains:

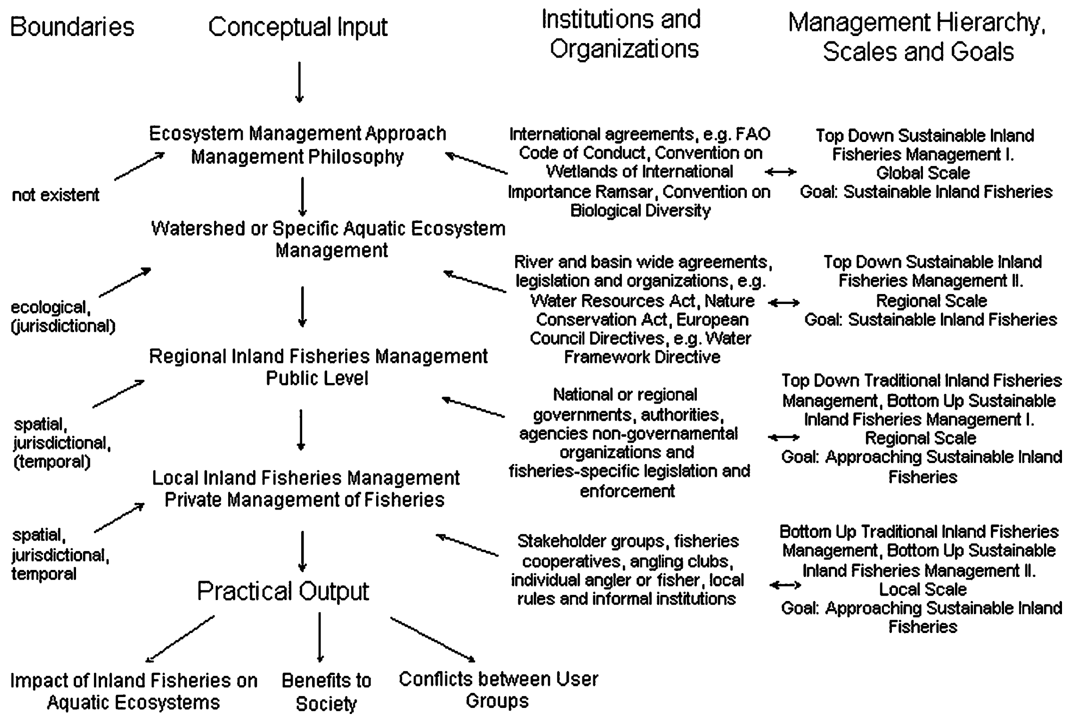

Figure 3 Management hierarchy of inland fisheries in Europe. The ecosystem management approach as 'new' management philosophy is rather the desirable than the existing condition.

environment, society and institutions (Pajak 2000; Fig. 4). Consequently, the interdependent components of sustainability in fisheries are ecological, social (socioeconomic and community) and institutional sustainability (e.g. Charles 1994, 2000). Unfortunately, in many scenarios these three domains (including scientific research) are disconnected and sustainability is compromised (Meffe 2002). Fisheries systems are sustainable if the aquatic ecosystems including their functions, services (e.g. recreation and nature conservation areas, see Holmlund and Hammer 1999 for details) and fish stocks persist in the long term (Costanza and Patten 1995). Such SD of inland fisheries conserves (land) water, genetic

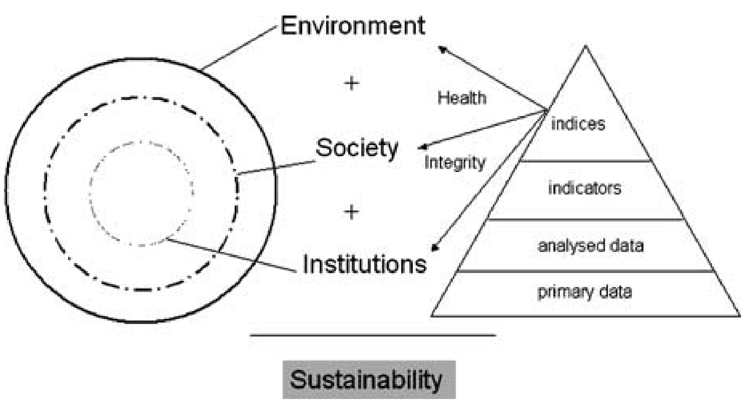

Figure 4 Sustainability domains of inland fisheries management and the potential role and use of sustainability indicators and indices as a communication tool to simplify and quantify information. The three domains of sustainability (environment, society and institutions) are illustrated as concentric rings, rather than partially overlapping, to more clearly depict their functional relationships and known hierarchy (societies and their institutions exist within the limits of their environments) (from Pajak 2000; modified).

resources, is environmentally nondegrading, technologically appropriate, economically viable and socially acceptable (FAO 1995).

Because of the inability to treat fisheries in isolation from the environment (Charles 1998), fisheries management systems have recently focused on ecosystem-based management (EBM, see Christensen *et al*. 1996) as the way to achieve sustainability (Larkin 1996; Schramm and Hubert 1999; Fluharty 2000). The EBM approach, as a philosophy in inland fisheries (Fig. 3), has evolved because of: (i) the recognition of the interrelationships of (eco)systems in time and space; inland fisheries are especially affected by land use practices – the inability of inland fisheries authorities to alter human activities on land ecosystems has led to focus on single-species management driven by sport and commercial fishing interests (Schramm and Hubert 1999); (ii) the consideration of human values in the process of natural resource management; human values are necessary for setting policy, establishing laws and ultimately making management decisions and actions (Cambray and Pister 2002) and (iii) the biodiversity crisis – biodiversity can only be conserved by rehabilitation or conservation of crucial aquatic habitats (Cowx and Collares-Pereira 2002; Cowx 2002a).

Despite widespread recognition and adoption of the general goal of sustainability and the EBM approach, most decision makers still lack a comprehensive set of management principles and an operational framework to monitor and manage for sustainable fisheries (Pajak 2000). Pajak (2000), Charles (2000) and Garcia and Staples (2000) developed a framework for fisheries which integrates a small set of critical, measurable components (sustainability indicators) necessary to maintain the integrity of ecosystems, societies and institutions (Fig. 4). Although not exhaustive, the proposed sustainability indicators (see the cited references for details), which during analysis are aggregated into sustainability indices (Fig. 4), have potential and should be developed further into a sustainability reference system (Garcia and Staples 2000) for inland fisheries management at higher levels of the management hierarchy (Fig. 3, see Section 4). Theoretical guidelines for management of inland waters are also available (Raat 1990; Welcomme 1998b, 2001), as well as a conceptual sustainability (policy or institutional) framework for fisheries, in the Code of Conduct of Responsible Fisheries of the Food and Agricultural Organization of the United Nations (FAO 1995; see also Caddy 1996, 2000; Garcia *et al*. 1999). Together with the technical guidelines for inland fisheries (FAO 1997), qualitative models (Welcomme 1999) and multidisciplinary/multivariate methods for appraisal of the health of fisheries (Pitcher *et al*. 1998; RAPFISH Pitcher and Preikshot 2001), all players of an inland management system should have sufficient 'global' input to develop efficient management systems 'locally'. Note, however, that maintaining a particular inland fishery in the long term (sustainably) may not always mean that sustainability in general or at higher levels (watershed, inland fisheries system in general) is achieved. It will therefore be essential to remain aware (and to identify) of the causes of unsustainability coming from outside the sector (e.g. land-based or aerial pollution), as well as those imposed by the inland fishery on other sectors (e.g. tourism, navigation) (Garcia and Staples 2000).

Some principles seem to be crucial for sustainable management of inland waters. These are: (i) responsibility; (ii) scale matching; (iii) precaution; (iv) adaptive management; (v) full cost allocation and (vi) participation (see Costanza *et al*. 1998 for details). These principles are not unique to inland fisheries but are basic guidelines governing the use of all environmental goods (Costanza *et al*. 1998). Thus, sustainable inland fisheries management systems will certainly require: (i) precautionary approaches and principles (e.g. Garcia 1994; FAO 1996; Richards and Maguire 1998; Auster 2001; Essington 2001; Hilborn *et al*. 2001); (ii) adaptive management systems (e.g. Walters 1986; Halbert 1993; Bundy 1998); (iii) participation of all stakeholders (e.g. Mace 1997; Charles 1998; Harris 1998; Pauly *et al*. 1998a; Sutton 1998; Cochrane 2000) and (iv) appropriate science and integrated (across disciplines, stakeholder groups and generations) management and approaches (Holling 1993; Rosenberg *et al*. 1993; Parrish *et al*. 1995; Stephenson and Lane 1995; Harris 1998; Policansky 1998; Caddy 1999). Besides, a reversal of the burden of proof must be applied to manage freshwater fishery resources so that those exploiting them must demonstrate no ecologically significant long-term changes (conservation-first perspective, precautionary principle) (Charles 1998; Dayton 1998). However, the key element of a new inland fisheries management system is a stronger participatory management process in which all stakeholders are represented (e.g. Decker *et al*. 1996; Garcia and Grainger 1997; Charles 2000; Iyer-Raniga and Treloar 2000). One essence of sustainability is the fisheries community view that they have not only the right to manage the resource properly, but also the duty (Walters 1998). This is referred to as a bottom-up approach to management and could be implemented as a cooperative management system (comanagement) where the fishing community manage the resource according to well-defined regulations with support from the appropriate governmental agency, or community-based management where the local stakeholders take direct control of the resource allocation and exploitation (see Pinkerton 1994; Sen and Raakjaer Nielsen 1996; Brown 1998; Costanza *et al*. 1998; Welcomme 2001; for details). The involvement of the communities in the management process is important because sustainable inland fisheries management will only be successful if: (i) all interest groups are willing to comply (e.g. Hønneland 1999); (ii) commitment of users to long-term conservation is assured (Sutton 1998) and (iii) conflicts between user groups are minimized or resolved through compromise solutions (Cochrane and Payne 1998; Cowx 1998a; Haggan 1998; but see Scheffer *et al*. 2000 for suboptimal compromise solutions from the overall social point of view). This needs more sophisticated education programmes and public outreach, which are of paramount concern in successful EBM (Holland 1996). Furthermore, effective communication skills are necessary to bring together a heterogeneous group of stakeholders and allow effective inland fisheries management (Brown 1996; Decker and Krueger 1999). Last but not the least, responsible (i.e., sustainable in the FAO Code of Conduct) management requires setting unambiguous and quantifiable objectives and management measures in cooperation with fishers, anglers and other interest groups (Lackey 1979; Barber and Taylor 1990; Cochrane 2000).

Sustainable inland fisheries will only become reality by: (i) taking into account the above principles; (ii) adoption of the EBM as management philosophy and (iii) implementing integrated, holistic, regional or local watershed/specific ecosystem management systems (which are beyond the scope of this paper). In addition, there is an urgent need to alter the current traditional management practices as prerequisites for sustainable fisheries (Fig. 3). It is easier to discard unsustainable management measures (Jamieson 1998) than to identify sustainable ones or even predict and define when sustainability is achieved ('after the fact', Costanza and Patten 1995). To understand better the main issues affecting inland fisheries in developed countries, these fisheries (Section 3) and their management practices (Section 4) are presented in more detail in the following two sections.

Description of inland fisheries of developed countries

This section describes the inland fisheries system of developed countries by: (i) defining the systems; (ii) summarising the benefits and negative impacts of inland fisheries and (iii) identifying impacts and threats to inland fisheries systems.

Definition and brief description

Generally, inland fisheries systems worldwide comprise four main categories: (i) commercial, capture food fisheries; (ii) noncommercial fisheries exploited for leisure, 'sport' or subsistence; (iii) aquaculture and (iv) upstream or downstream services such as gear manufactures, ownership of water rights, tourism (upstream) and fish processors, transporters and retailers (downstream) (Steffens 1986; O'Grady 1995; Welcomme 2001; Cowx 2002c). Aquaculture is often treated separately from the inland fisheries sector as it is more akin to an agricultural activity (FAO 1997). The demarcation between capture fisheries and aquaculture is becoming increasingly difficult to define, as fisheries management is mainly achieved through enhancement activities, such as stocking, fertilization and environmental manipulation (Welcomme and Bartley 1998; Welcomme 2001). However, there are important ecological, sociological and legal differences between capture fisheries and aquaculture (Welcomme 2001). Thus, for the purpose of this paper, inland fisheries are defined as fishing activities in natural or 'semi natural', limnetic ecosystems, such as rivers, lakes, gravel pits, other man-made standing water bodies and reservoirs, to benefit from the use of fish and other aquatic organisms therein.

In commercial fisheries, static (e.g. gill nets, long lines, taps, fyke or bag nets) and active gears (e.g. seine nets, electric fishing, cast nets, lift nets) are used (see Welcomme 2001 for details). Fishing with trawls is not common in inland waters of most developed countries (but compare Jurvelius and Auvinen 2001 in Finland) and often prohibited by law, as are the use of poisons and explosives. Full-time commercial food fisheries have largely disappeared throughout the developed world (e.g. North America, Belgium, UK). Locally, however, there have remained economically viable (but threatened) commercial capture enterprises in some European countries (e.g. Germany, Finland, France, Poland, see Von Lukowicz 1995; Bninska 2000; Salmi *et al.* 2000; Boisneau and Mennesson-Boisneau 2001; Wedekind *et al.* 2001; for details). In these countries, the sole surviving (mostly part-time, Salmi *et al.* 2000; Rasmussen and Geertz-Hansen 2001; Arlinghaus *et al.* 2002) inland fishermen often operate multidimensionally as service industries providing recreational activities (fishing, boating, swimming), managing fish communities, processing (e.g. smoking) their landings and supplying them directly to consumers, restaurants or hotels. Commercial enterprises in developed countries are often highly dependent on anglers who buy fishing licences or rent boats, especially in regions with tourism activities or close to metropolitan centres. Thus, most of the surviving commercial fishermen are directly or indirectly linked to recreational fishing (Wortley 1995), and so are the aquaculture facilities that often provide stocking material for fishery enhancements (Wortley 1995; FAO 1997). A reduction in the numbers of anglers would mean a reduction of the whole interdependent inland fisheries sector.

By far, the most important branch of the inland fisheries sector in developed countries is recreational fishing using rod, reel, line and hooks (angling), or a variation of them (Guthrie *et al.* 1991; Pollock *et al.* 1994; Aas and Ditton 1998; Welcomme 2001; Cowx 2002c). In some countries (e.g. Finland, Norway), recreational fishing is practised with gears which are predominately designed for commercial purposes (e.g. gill nets, traps) (Aas and Skurdal 1996; Salmi *et al.* 2000). Thus, although recreational fishing is usually considered a leisure activity (Aas 2002), in poorer countries and some more prosperous ones such as Finland or Germany, recreational fisheries play a role in supplementing diets and most fish caught are taken home for consumption (Bninska 1995; Welcomme 2001; Cowx 2002c). In such circumstances, gillnets can be an important fishing gear for a type of fishery that is not commercial and not recreational *per se*, but where fishing is used as a means to supplement incomes and direct consumption of food (subsistence fishing, Glass *et al.* 1995). However, in many areas the distinction between subsistence and recreation is unclear, and especially in developing countries and rural districts it is necessary to consider subsistence and recreational fishing together (Aas and Skurdal 1996; Aas 2002). Thus, in this paper the term 'recreational fishing' includes all the fishing practices which are noncommercial including angling and gill-netting harvesting fish for subsistence.

Social sciences revealed great angler diversity in terms of social and economic characteristics, motivations, leisure benefits sought from fishing, participation patterns, species and method preferences, levels of involvement and commitment, consumptive orientation and management preferences (e.g. Hahn 1991; Ditton 1996; Aas and Ditton 1998). The 'average' angler exists only in research reports (Aas and Ditton 1998). The concept of recreational specialization provides a means for understanding the diversity of anglers (see Hahn 1991 for a review). Bryan (1977) defined specialization as 'a continuum of behaviour from the general to the particular reflected by equipment and skills used in the sport and activity setting preferences'. Bryan (1977) identified four types of trout anglers: occasional anglers (lower end of the specialization continuum), generalists, technique specialists and technique and setting specialists (upper end). Another possibility of segmenting anglers is according to their consumptive orientation (Fedler and Ditton 1986; Aas and Kaltenborn 1995). Degree of consumptiveness should be seen as an orientation to value consumptive (catch-related) over nonconsumptive motives (noncatch-related) for angling. Welcomme (2001) and Cowx (2002c) proposed another broader method for segmentation of anglers into: (i) match anglers (competition anglers); (ii) specimen anglers (trophy anglers practising catch-and-release or 'game' angling for larger salmonids); (iii) relaxation anglers and (iv) anglers fishing primarily for food. However, many anglers are difficult to group because they often pursue more than one type of fishing (Hahn 1991; Pollock *et al.* 1994).

Numerous studies confirmed that a great percentage of anglers seek to escape from daily routines and relax in nature and are not necessarily concerned about the quantity or quality of their catch to enjoy their fishing trip (see Fedler and Ditton 1994 for a review). In recent years, however, in Europe there has been a trend towards more catch-related motivations, and to specialize on certain species (Cowx 2002c). An increasing trend in recreational fisheries, which has been practised in countries like the UK and France for decades, is for the fish, once captured, to be returned to the water for the capture of others or the conservation of fish stocks (Policansky 2002). Catch-and-release policies are fundamental to the recreational fisheries policies of the USA and other temperate countries such as the Netherlands and the UK (Welcomme 2001; Policansky

2002). However, this procedure has been questioned by animal welfare groupings in several countries, such as Germany, where the reasonable reason for harming an animal is often only accepted if the catch is removed for consumption (Berg and Rösch 1998).

With respect to management of inland waters in developed countries, stocking of fish coupled with habitat rehabilitation measures have become the major actions (Table 1; Welcomme 2000, 2001; Cowx 2002b). Declining quality of the aquatic environment, resulting from eutrophication, pollution, acidification and habitat modification, coupled with poor fisheries management and the creation of a popular stocking mythology among fisheries managers and fishery stakeholders (Meffe 1992), has over the past century led to an emphasis on enhancement of fisheries to mitigate anthropogenic stress (Schramm and Piper 1995; Cowx 1998b). Today, most inland fisheries are stocked freshwater fisheries, so-called culture-based fisheries (FAO 1997; Cowx 1994b, 1998b, 2002b; Welcomme and Bartley 1998; Welcomme 2000). The main types of fisheries based on stock enhancement, as opposed to natural fisheries without stock enhancement practices, are: (i) put, grow and take; (ii) put and take and (iii) put, catch and release (see Cowx 2002c for details). In Europe and in particular in the UK, there is a growing trend towards the latter type of recreational fishery. These fisheries are often small, highly stocked stillwater fisheries providing anglers with eating facilities, rest rooms, bait and tackle suppliers and entertainment for the whole family (Lyons *et al*. 2002). The angling experience gained from these intensively managed, purpose-built artificial fisheries offering easy access and high catch rates (North 2002) is considerably different from the angling experience in natural fisheries. However, artificial fisheries may ease the angling pressure from natural water bodies at the expense of increasing angler catch expectations which can no longer be satisfied in natural water bodies.

Benefits – values and impacts

Inland fisheries undoubtedly have high socioeconomic and sociocultural importance and provide 'a myriad of benefits to society' (Fisheries Management and Ecology, 2001; Weithman 1999, 2001; Pitcher and Hollingworth 2002). However, benefits created by inland fisheries are difficult to group, quantify and evaluate (e.g. Talhelm and Libby 1987; Kearney 1999, 2002). This is particularly true for recreational fisheries (Cowx 1999a, 2002c). However, because people go fishing, despite the time and money required, indicates that there are values associated with participation. To an angler, a fishing trip is an experience that includes such dimensions as relaxation or escape from work-related pressures, friendship, enjoying out-of-doors, challenge, and the opportunity to consume the fish that are caught. A fishing trip has a planning phase and a recollection phase, as well as the event itself. Each of these phases is generally viewed positively by anglers and therefore has benefits that accrue to anglers (Pollock *et al*. 1994). However, an important distinction has to be made between two types of benefits: values and impacts (Probst and Gavrilis 1987; Weithman 1999). Values describe what people receive related to their expenses – for anglers, the satisfaction of the trip; for commercial fishers, profit (Weithman 1999). It should be recognized that value in this context refers to the benefits accrued by the angler and not the capital value of the fisheries which can be substanial, especially for salmon fisheries. Impact, in contrast to values, represents the effects that are generated by the use of the resource, i.e. effects to the community and local, regional and national economies (Weithman 1999). Concerning socioeconomic impact, the distinction between users and beneficiaries of the resource use is crucial (compare Fig. 2). In commercial fisheries, relatively few people use the resources through fishing, but many people benefit from the fish that is bought for consumption. In recreational fisheries, irrespective of leisure or subsistence or a combination of both, a larger number of people fish (use), when compared with commercial fisheries, but there are not many beneficiaries over and above these recreational fisher numbers. In nature conservation, few people use the resource directly for aesthetic reasons (Smith 1986), but the general public benefits by an improvement in ecosystem health, albeit reluctantly.

Generally, three domains can be distinguished where benefits are accrued, viz. economic, social and ecological benefits of inland fisheries. Furthermore, when speaking about impacts additional components need to be taken into account: (i) negative impact of inland fisheries on aquatic ecosystems and (ii) impacts, threats and constraints on inland fisheries.

Economic benefits

The most transparent benefit of inland fisheries is an economic one. There are two types of economic

Table 2 Socioeconomic benefits of inland fisheries and impacts on inland fisheries (from Weithman 1999; modified).

Values		Impacts
	Economic benefits	
User: Consumptive, nonconsumptive, indirect		Direct, indirect, induced
Nonuser: Option, existence, bequest		
	Social benefits	
Cultural, societal, psychological, physiological		Quality of life, social well being
	Ecological benefits	Mitigation, rehabilitation, management, negative 'benefits' (impacts)
		Other impacts
		environmental degradation, low societal priority, user conflicts, cost-effectiveness, constraints

values associated with fisheries: user and nonuser, whereby a user, such as an angler, off-site of a particular water is by definition a (temporary) nonuser of the fishery. User value can be subdivided into consumptive, nonconsumptive and indirect values (Bishop *et al.* 1987; Randall 1987; Table 2). Consumptive use is, for example, harvest by an angler. Nonconsumptive uses of fisheries resources include sightseeing, and enjoyment of nature, fresh air and other public goods that do not deplete the fishery resources, i.e. the value that individuals derive that is not conditional on consumption of, or physical change to, the natural resources. On-site use includes consumptive and nonconsumptive activities associated with the fishery (Riechers and Fedler 1996). Indirect use may include activities away from the site, including reading about or special activities at the fishery location (Riechers and Fedler 1996).

Nonuse value is partitioned into option (value to an individual of maintaining the option to use a resource some time in the future), bequest (value to an individual knowing that a resource is available for future generations to use) and existence value (value derived by an individual from knowing that a resource exists and that others have the opportunity to use it) (e.g. Loomis and White 1996; Riechers and Fedler 1996; Weithman 1999; Peirson *et al.* 2001; Table 2).

Use and nonuse values can be determined through contingent valuation methodologies (CVM) (e.g. Navrud 2001a). These involve asking a sample of the relevant population either (i) how much they are willing to pay (WTP) for a service or increase they are willing to pay for maintaining access to the service or (ii) how much they are willing to accept (WTA) as a compensation for a loss of the service or a change not occurring. Since it is often improvements in the quantity or quality of fish stocks that are being assessed, the appropriate measure is either compensation surplus (WTP for improvement) or equivalent surplus (WTA for the change not occurring) (see Perman *et al.* 1999 for discussion). The choice of WTP or WTA depends on assumptions about entitlements and whether the change is an improvement or a deterioration in environmental quality. Generally, WTA is only used where there are clear property rights to the status quo and the change is a deterioration (Peirson *et al.* 2001). WTP, which includes actual expenditures and excess value (benefits that exceed monetary cost, net economic value or consumer surplus) to users, is an appropriate measure of economic value of a recreational fishery (see Pollock *et al.* 1994; Riechers and Fedler 1996; Weithman 1999; Navrud 2001a for reviews) and of a part-time or artisanal commercial or subsistence fishery, which are comparable to 'leisure' activities. In addition, the value that nonusers place on recreational fisheries has to be considered if total economic value of a recreational fishery is to be evaluated.

In addition to the CVM, other nonmarket valuation techniques, such as the travel cost method, have been developed to determine the economic value of recreational fishery resources (e.g. Pollock *et al.* 1994; Navrud 2001a). In Europe, however, such economic valuation has only been applied to recreational fisheries in selected parts, e.g. the UK (Postle and Moore 1998; Peirson *et al.* 2001) and Scandinavia (Sipponen 1990; Navrud 2001a; Roth *et al.* 2001; Toivonen 2002). Instead, in many countries (e.g. Germany), there has been a focus on economic impact assessments (Probst and Gavrilis 1987) to 'value' the fishery. Unfortunately, expenditure and economic impact data do not fully account for the economic value of a recreational fishery (Edwards 1991; Fisher and Grambsch 1991; Riechers and Fedler 1996). For inland fisheries conservation as well as management, it is essential to provide a thorough economic evaluation of inland fisheries to defend the position of the sector *per se* against aquatic resource development schemes (Cowx 1998a, 1999a, 2002a,c). Some examples of WTP or net economic value of recreational fisheries are given below.

Net economic value of recreational fishing varies considerably depending on the particular country, type of fishery and the question format and statistical model of the valuation study. In a multinational fisheries survey in the Nordic countries (Toivonen 2002), mean additional WTP of recreational fishers per year (converted into US$ by 1999 purchasing power parties) was estimated, with US$56.5 in Sweden, US$71.3 in Denmark, US$73.2 in Finland, US$82.4 in Norway and US$139.8 in Iceland. Mean nonfisher's WTP per year for recreational fishing ranged from US$47.1 in Finland to US$133.1 in Iceland and was not considerably lower than the user's WTP. This indicates that nonusers can place a high (nonuse) value on recreational fishing. Aggregated nonuse value, e.g. nonuser WTP, can make up a major part of the total economic value of aquatic ecosystem services (Baker and Pierce 1997; Navrud 2001a).

Net economic value of recreational fishing can reach very high values. In and near Yellowstone National Park (USA), anglers placed a value of between US$172 and 977 (2002 dollars) on a day of fishing (Kerkvliet *et al.* 2002). The net economic value of angling in Alaska was estimated with US$285 per day (1997 dollars) (Duffield *et al.* 2002) and the consumer surplus of an angling trip in the Costa Rican billfish recreational fishery was estimated with US$1777 per trip (Rudd *et al.* 2002; see Pitcher and Hollingworth 2002 for more estimates). Lower WTP values were reported in other studies. In USA, the mean net economic value of a day of cold-water fishing (in 1987) was estimated at US$30.62, anadromous fishing at US$54.01, warm-water fishing at US$23.55 and salt-water fishing at US$72.49 (see Walsh *et al.* 1992 for confidence intervals). Road-accessible stocked waters in the Fairbanks area (Alaska) had per day values ranging from $15 to 44 (Duffield *et al.* 2002). In the UK, the value of angling for coarse (nonsalmonid) fish for a day in rivers was estimated (in 2002) to be US$7.1–12.5 (Lyons *et al.* 2002), and the recreational value per angling day for salmonid fishes in Norway was reported to range between US$4.4 and 7.15 in saltwater and between US$4.84 and 35.31 in fresh water (CVM estimates in Navrud 2001a).

These WTP or consumer surplus estimates can be used in benefit–cost analysis to evaluate the recreation benefits of improvements of environmental quality in relation to the economic losses (costs) for other water uses such as irrigation or hydropower generation. Willis and Garrod (1999), for example, investigated the benefits to anglers and other recreation users (e.g. swimming, wildlife viewing) of increasing flows along low-flow rivers in England. Mean WTP of anglers was £68.03 (≈US$95.2) per year for improved fishing brought about low-flow alleviation. Mean WTP for other on-site recreationists was £28.22 (≈US$39.5) per year and for nonusers (informal recreationists of the general public) the mean WTP ranged between £5.34 and 10.78 (≈US$7.5–15.1) per year for an environmental acceptable flow regime. Willis and Garrod (1999) demonstrated that the benefits to anglers alone outweighed the costs of low-flow alleviation programmes in two of the seven rivers evaluated in south-west England. The value of other recreationists and nonusers justified the low-flow alleviation in another three rivers. Only where the costs of low-flow alleviation were extremely high did recreational benefits fail to exceed the costs of implementing an environmentally acceptable flow regime in the investigated rivers. Other studies also demonstrated that marginal increases in stream flow can generate benefits to recreational fishing that exceed the marginal value of water in agriculture (Hansen and Hallam 1991). However, there might also be net losses associated with a change in management regimes which benefit outdoor recreation including fishing, but constrain commercial enterprises such as hydropower generation. Analysing the costs (reduced production of electricity) and benefits (increased recreation value) of different high water level management alternatives in four North Carolina reservoirs, Cordell and Bergstrom (1993) found a net economic loss under several scenarios. In contrast, other cost–benefit analyses revealed that social benefits in terms of increased use (angling) and nonuse values to be several times higher than the costs of rehabilitation of habitats (e.g. liming and restocking acidified rivers and lakes in Norway; Navrud 2001a).

Willis and Garrod (1999) demonstrated that WTP of recreational fishers can be higher than the WTP of nonusers or other recreational users (e.g. wildlife viewers). Creel and Loomis (1992), on the other hand, found that per participant use value of recreation at

14 recreational resources in the San Joaquin Valley (USA) was higher for wildlife viewers (US$128–152 in 1989 dollars) and waterfowl hunters (US$149–159) than for anglers (US$126–137). However, total recreation benefits (million US dollars) were similar for wildlife viewing (US$37–44 million) and fishing (US$32–34 million), and considerably higher than for hunting (US$7 million). In a literature review, Walsh *et al*. (1992) reported lower mean net economic values per day of camping (US$19.5), picnicking (US$17.33), swimming (US$22.97) and nonconsumptive fish and wildlife activities (US$22.2) than for various fishing types (US$23.55–72.49 million). Conversely, the net economic value per day of wilderness activities ($24.58), hunting (US$30.82–45.47) and boating (US$31.56–48.68) was in the range of values of a fishing day. The described relationships have several implications: First, nonfishing stakeholders may place at least the same value per individual on the resource as recreational fishers. Second, aggregate value of resource use or nonuse by different stakeholders is dependent on stakeholder numbers, which in the case of nonfishery stakeholders may outweigh a higher mean WTP of recreational fishers compared with other water uses. Third, dependent on the particular situation, cost–benefit analysis using recreational fishing value can demonstrate that improvements of environmental quality may be very profitable investments. Fourth, the high variability of net economic value estimates of recreational fishing limits the applicability of benefit transfer which is the process by which researchers take recreational value estimated for one site or region and apply them to another site or region (Walsh *et al*. 1992). Therefore, there is a need to value as many types of recreational fisheries and locations as possible.

Profit through the provision of animal protein to society is a useful measure of economic value of a commercial fishery because, like consumer surplus, profit is value in excess of costs (Edwards 1991). However, commercial fishermen experience certain value components, which are not embraced by profit alone (Lackey 1979; Hart and Pitcher 1998), e.g. producer surplus (Edwards 1991). Irrespective, without profit a commercial enterprise would leave the fishery unless it is subsidized. Net economic value of commercial fishing comprises consumer and producer surplus, the latter of which is not quite equivalent to profit (Edwards 1991). Because there are market prices in commercial fisheries, demand and supply functions allow determination of economic value of commercial fishing. Care has to be taken to compare revenues or profits of commercial fisheries with economic value of recreational fisheries to allocate fishery resources because these 'economic arguments' derive from fundamentally different economic concepts (see Edwards 1991 for critique). Instead net economic value or consumer surplus of recreational fisheries and net economic value of commercial fisheries which is consumer and producer surplus should be compared, and allocation be based on the basis of incremental tradeoffs in net economic value (see Edwards 1991 for details).

Expenditure by anglers or commercial fishers represents revenues and jobs generated in local economies. There are three types of economic impacts: (i) direct impacts, which are the purchases made by fishermen, including travel, accommodation and food costs; (ii) indirect impacts, which are the purchases made by businesses to produce goods or services demanded by fishermen and (iii) induced impacts, which are the purchases of goods and services by households receiving wages from businesses producing direct or indirect goods. The summation of these three levels of impact is the total economic impact (TEI). TEI divided by the direct impact is called the multiplier and reflects the number of times the initial expenditure circulates through the local economy (from Riechers and Fedler 1996; compare also Edwards 1991). Regional economic impact studies have been conducted of recreational fishing in both fresh water (e.g. Schorr *et al*. 1995) and salt water (e.g. Bohnsack *et al*. 2002). For example, the impact analysis for planning (IMPLAN) modelling system evaluated the effect of fishing expenditures (US$25.641 million in 1990) on the regional economy bordering Lake Texoma (USA) (Schorr *et al*. 1995). Direct, indirect and induced impacts of these expenditures were directly associated with US$57.392 million in total business sales, US$23.273 million in value added; and 718 jobs in the impact region.

Social benefits

Brown and Manfredo (quoted in Weithman 1999) identified four categories of social value in inland fisheries: cultural, societal, psychological, and physiological (Table 2). The former two pertain more to nations and regional communities, whereas the latter two relate to individuals (Weithman 1999; see also Kearney 2002).

Cultural values represent a collective feeling toward fishes and fishing. Fishing in inland waters is an important societal asset and is valued by the community as a whole. Societal values are based on relationships among people as part of a family or community (e.g. family fishing). Psychological values are those that relate to satisfaction, motives or attitudes associated with the use, or knowledge of the existence, of a fishery. Physiological values relate to improvements in human health (e.g. reduction of stress) related to fishing (Weithman 1999).

Social impacts are very elusive (Vanderpool 1987). They relate to quality of life and social well being caused by fishing (Gregory 1987). For example, attracting lots of anglers to a particular well-managed, high-quality lake would generate income to the commercial fishing community and increase social well being, which can be measured through improved quality of life.

Ecological benefits

Ecological benefits of inland fisheries are, typically, difficult to quantify (Kearney 1999, 2002; Table 2). Because of the impairment of most fresh waters of the northern temperate world, there is an increasing trend towards intervening either to improve the functioning of degraded systems or to restore them (Cowx 1994a; Cowx and Welcomme 1998; Welcomme 2001). In Europe, this is sometimes transformed into practice by inland fishermen or angling clubs who are traditionally the stakeholders charged with the practical management of freshwater fisheries (Fig. 3, see Section 4 for details). Thus, many inland fisheries management activities aim to mitigate or rehabilitate the adverse human-induced changes by manipulating the ecosystems in an attempt to gain positive benefits. The duty to manage freshwater ecosystems is often promulgated in fisheries laws (e.g. Germany, UK), making managers of inland fisheries and fishery stakeholders key players for implementing EBM. Kearney (1999) suggested that the conservation-conscious fishing community represents one of the greatest potential forces for the conservation of aquatic biodiversity. Kearney (2002) further stressed that recreational fisheries have different potential positive ecological impacts such as education, promotion of environmental responsibility, aid in environmental monitoring, engendering support for restoration and aid of surveillance of environmental vandalism. Irrespective, nonfishery specific international agreements (e.g. European Habitats Directive 1992), institutions (e.g. nature conservation acts, water resources acts), public authorities (e.g. environmental, nature conservation or river engineering agencies), and various stakeholder groups and nongovernmental organizations (NGOs), including the public (e.g. nature conservation groupings), usually have a greater role to play in conservation of freshwater (fish) resources on a larger scale when compared with fishery stakeholders (e.g. Cowx 2002a; Kirchhofer 2002; Schmutz *et al.* 2002, Fig. 3). However, inland fisheries players have the potential to become strong allies for and provide constituency support for EBM approaches and to change environmentally unfriendly behaviour of fishery managers, anglers and fishers which impact negatively on ecosystem structure and function (see below) (Cambray and Pister 2002). Indeed, many traditional regulations of inland fishing management systems (see section 3), which were not enacted primarily for fish species conservation, made important contributions to species conservation (Kirchhofer 2002). Indirectly, in some countries, fishery stakeholders (e.g. angling societies) have pushed governments to formulate environmental legislation and were the driving forces for several subsequent legal revisions (Kirchhofer 2002). These developments, which, for example, in the case of a water protection act initiated by fishery associations in Switzerland in 1955, were milestones in fish and freshwater conservation.

Depending on the societal status of the inland fisheries systems within the communities of developed societies, the contribution of inland fisheries towards the protection and conservation of freshwater ecosystems is disregarded or admired. In Berlin (Germany), for example, commercial fisheries have been accepted as a major actor to improve water quality by reducing the biomass of planktivorous cyprinid fishes, i.e. biomanipulation (Grosch *et al.* 2000). With the help of anglers' expenditure, many management programmes have been initiated to conserve stocks of threatened riverine fish species such as Atlantic salmon (*Salmo salar* L., Salmonidae) (VDSF 2000) or migrating brown trout (*Salmo trutta* f. *lacustris* L., Salmonidae) (Ruhlé 1996; Hughes and Willis 2000; VDSF 2000). In many cases, anglers' expenditure is often the driving force financing inland fisheries and aquatic ecosystem management (e.g. Radonski 1995; Ross and Loomis 1999; Rasmussen and Geertz-Hansen 2001; Kirchhofer 2002).

Recreational fishing clubs invest much effort, money and time to manage their angling waters. These waters often have restricted access, thus the ecosystems are well protected against environmental degradation or pollution, and club members serve as watchdogs against adverse influences (Lyons *et al.* 1999, 2002). There are also examples of anglers' conservation associations (e.g. Anglers' Conservation Association formed in 1948 in the UK) which were organized to fight through legal actions against environmental pollution (Bate 2001). However, not all measures adopted by the traditional inland fisheries management are considered positive. In some countries (e.g. Germany) where the contemporary environmental and animal rights movements are strong, ecological benefits of inland fisheries are not accrued. Moreover, common management measures such as stocking are sometimes considered the most serious threat to biodiversity of fish (Cowx 2002a; Freyhof 2002). Regardless of these potential negative impacts, a relatively high proportion of society keeps in contact with nature through linkages with inland fisheries and consequently tends to be more sensitive to environmental issues than the majority of an increasing urban population (Lyons *et al*. 2002). This awareness of environmental issues and diversity of ecosystems by fishing protagonists (e.g. Hammer 1994; Kearney 1999; Connelly *et al*. 2000) is paramount for EBM (e.g. Olsson and Folke 2001) and sustainability, assuming that ecological responsibility is achieved. Furthermore, indigenous knowledge of the fishing fraternity and informal (local) institutions can play an important role in the sustainable management of fishery resources (e.g. Long and Chappell 1997; Mackinson and Nøttestad 1998; Jentoft 1999; Berkes *et al*. 2000; Johannes *et al*. 2000).

Negative impacts on aquatic ecosystems

Inland fisheries also have negative ecological impacts on aquatic ecosystems (Table 2). The most direct impact is caused by exploitation of natural (renewable) resources and subsequent disruption of the natural processes. Changes in the composition and abundance of fish assemblages have resulted not only from the degradation of the environment, but also through actions to influence the structure of the fish community and through fishing itself (Welcomme 1992). It is well known that the fisheries of most inland waters undergo a series of changes in response to fishing pressures which force the assemblage towards smaller-sized species and individuals (e.g. Hoffmann 1995, 1996; Welcomme 1999). Externally induced stress leads to a replacement of large, long-lived species, usually with complex life histories (of the type characterized as 'K' selected or climax), by smaller, shorter-lived species with simpler life histories (corresponding to 'r' selected or primary colonizers) (Welcomme 1992, 1995, 1999, 2001). Environmental degradation tends to induce many of the same changes in assemblage structure as does fishing (Rapport *et al*. 1985). The relative weighting of these stresses varies according to the type of aquatic ecosystem. In many rivers, for example, major impacts appear to have occurred through environmental degradation (Welcomme 1992, 1999). In lakes, on the other hand, fish populations tend to be heavily influenced by direct manipulation of the assemblage through addition of desirable species, removal of unwanted species or overfishing (e.g. Regier and Loftus 1972; Welcomme 1992; Debus 1995; Cowx 2002a). However, cultural euthrophication as a result of urbanization has also had a detrimental effect on the species assemblage in lakes (e.g. Hartmann 1977; Persson *et al*. 1991; Cowx 2002a). This makes it very difficult to detect the main agent responsible for the degradation of freshwater ecosystems and inland fisheries (Welcomme 1992, 1997, 1999).

Generally, large fishes, which are sensitive to high fishing pressure, are piscivorous. The reduction in mean size of fishes as a result of over fishing correlates with a reduction in piscivorous fish species in favour of small-bodied planktivorous ones (Pauly *et al*. 1998b). This shift caused by fishing has been called 'fishing down food webs' (i.e. at lower trophic levels) (Pauly *et al*. 1998b). This effect is most pronounced in the Northern Hemisphere (Pauly *et al*. 1998b). However, in inland waters, stocking and other anthropogenic impacts often counteract any 'fishing down' effect and thus precludes evaluation of overfishing and sustainability based on mean length of fishes (Pauly *et al*. 2001; Post *et al*. 2002). However, it has long been suggested that overfishing reduces the heterozygosity of target species by the selective elimination of individuals that are more readily captured (e.g. fast growing individuals of piscivorous species; Welcomme 1992; Policansky 1993; Ratner and Lande 2001), which may cause evolution of the community to slower growing fishes (Favro *et al*. 1982). Furthermore, in freshwater fish populations the degree of exploitation influences the age at which fishes become mature (Healey 1978; Bowen *et al*. 1991), which can be reduced by high fishing pressure (McAllister *et al*. 1992; Rochet *et al*. 2000).

In some European countries (e.g. Germany, the Netherlands, Switzerland), piscivorous fish such as pike (*Esox lucius* L., Esoxidae), pikeperch (*Sander lucioperca* (L.), Percidae) and perch (*Perca fluviatilis* L., Percidae) have a substantially higher market value than zooplanktivorous fish such as roach (*Rutilus rutilus* (L.), Cyprinidae) or bream (*Abramis brama* (L.), Cyprinidae) (e.g. Von Lukowicz 1995,1997). Thus, piscivorous fish species are often selectively removed from the ecosystems by commercial fishermen (e.g. Bninska 1995; Von Lukowicz 1995; Grosch *et al.* 2000). In addition, recreational fishermen in many European countries target piscivorous fish, which are removed for consumption (Benndorf 1995; Bogelius 1998; Schwärzel-Klingenstein *et al.* 1999; Steffens and Winkel 1999; Rasmussen and Geertz-Hansen 2001; Arlinghaus 2002). Consequently, in countries where inland fishermen remove selectively 'top down' components of the food webs (piscivorous predators), inland fisheries can ultimately contribute to the deterioration of the water quality in eutrophic lakes through 'negative' biomanipulation.

One of the major impacts of inland fisheries on aquatic ecosystems is suggested to originate from fish stocking and introductions (e.g. Moyle 1997; Cowx 1998b). Stocking and introduction of fishes can be damaging to native stocks through mechanisms such as genetic contamination, hybridization, disequilibrium of fish populations, spread of disease, environmental disturbance, predation, competition and cointroduction of nuisance species (see Cowx 1994b, 1998b; Maitland 1995; Ryman *et al.* 1995; Schramm and Piper 1995; Welcomme and Bartley 1998; Welcomme 2001 for details and case-studies). Unfortunately, there is a surprising lack of (scientific) information on how stocked systems function, and on the benefits, successes and risks associated with stocking programmes (Cowx 1994b, 1998b; Welcomme 2001, see below for more details). Many ecologists agree that stocking practices to augment existing populations should only be performed with fish reared from the indigenous population because the latter should always contain the optimum genotypes for a particular locality (Elliott 1995). However, stocking has taken place over hundreds of years (e.g. Balon 1995); thus, it might be difficult to find autochthonous populations (Ryman 1991).

Fishing can have impacts on the abundance and composition of fish assemblages through mechanical damage provoked by fishing gear, particularly mobile gears such as trawls or seines (Welcomme 1992). Besides, fishing practices can be harmful to fishes (see Balon 2000; Rose 2002 for discussion). For example, mesh selective gears can damage fish through descaling if they escape from the net (Welcomme 1992). Electric fishing is known to sometimes damage fish (Cowx and Lamarque 1990; Reynolds 1996). In recreational fisheries, catch-and-release fishing can cause hooking mortality (Munoeke and Childress 1994) and have effects on behaviour, growth or reproduction of fish (Policansky 2002). Furthermore, wading recreational fishermen can damage spawning sites of fishes in low order rivers.

Other impacts of activities linked to inland fisheries on the ecosystems are conceivable. Presence at the waterside can disturb wildlife, such as waterfowl, birds, or mammals (Cowx 2002c). Litter, such as fish hooks, lead and plastics, not only reduces aesthetic beauty, but also is suspected of damaging wildlife (Bell *et al.* 1985; Cryer *et al.* 1987). The practice of ground- and pre-baiting (e.g. maggots, cereals and, boilies – a carp (*Cyprinus carpio* L., Cyprinidae) bait) is common to attract fish in recreational nonsalmonid fishing (Cryer and Edwards 1987). When used excessively, it can contribute to anthropogenic eutrophication (Arlinghaus and Mehner, unpublished data) and lead to a substantial reduction in benthic fauna (Cryer and Edwards 1987). The ethical considerations of harming vertebrates by catch-and-release angling or by keeping fish alive for bait in keep nets or buckets is a major source of debate among animal welfare and rights groups on the one hand and inland fishermen and their lobby groups on the other (Maitland 1995; De Leeuw 1996; List 1997). However, recent studies on holding fish in keep nets suggest that the fish are not unduly stressed until the density held is high (Pottinger 1997; Raat *et al.* 1997).

Riparian vegetation and ecotones (e.g. Schiemer *et al.* 1995) can also be damaged by fishermen gaining access to the water or operating active gears leading to an increased nutrient inflow (Bninska 1985). All measures which can be broadly considered as habitat management, such as construction of groynes or fishing platforms, clearance of aquatic weed, etc. (see Cowx and Welcomme 1998 for details), are potentially damaging to the environment (Maitland 1995). There is also debate (Cowx 1999b, 2002a) about whether rehabilitation practices are beneficial to ecosystem functioning. For example, Van Zyll de Jong *et al.* (2000) and Amisah and Cowx (2000) showed that the expected outcomes of rehabilitation action were not always achieved and opposing, often conflicting, outputs were accrued. This is not a common end product of rehabilitation activities, but lack of emphasis on post project evaluation results in poor documentation of the adverse effects of so-called 'improvement works' (Cowx 1999b).

To conclude, fishing activities have different negative impacts on aquatic ecosystems that need to be addressed by sustainable inland fisheries management systems. However, worldwide the main causes responsible for freshwater fish species decline and extinctions (in 71% of all known species extinctions) are various forms of habitat alterations (Williams *et al.* 1989; Harrison and Stiassny 1999). Of all the rare North American fish species, only 11 taxa (3%) were listed, at least in part, because of losses from overuse for fisheries, scientific or educational purposes (Williams *et al.* 1989), and the large-scale biodiversity decline in the southern USA is a vital signal of the pervasive nonfishing-induced degradation of aquatic ecosystems and watersheds (Warren *et al.* 2000). However, some fishery management activities, especially introduction of (exotic) species, have resulted in species extinctions (Cowx 1996, 1998b; Moyle 1997) and were involved in more than half of all recorded fish species extinctions worldwide (Harrison and Stiassny 1999). Overall, inland fisheries seem to contribute less than other manufacturing and chemical industries or even agriculture to the problem of global change and biodiversity degradation (Garcia and Grainger 1997). Furthermore, most fisheries' effects (e.g. overharvesting), except harmful introductions and translocations, are reversible, while degradation of critical habitats is often not (Garcia and Grainger 1997). Nevertheless, many freshwater fish species extinctions seem to involve complex synergisms. For example, Harrison and Stiassny (1999) found that in many cases habitat modification, pollution and introduction of exotic species or overfishing appear to have worked either simultaneously or sequentially to bring about the eventual extinction of the species. Examples are the apparently extinct or threatened salmonid species of *Coregonus* and *Salvelinus* in the postglacial European lakes and the North American Great Lakes. These populations that might normally have withstood competition from introduced species or the effects of pollution have been unable to do so because their numbers were already weakened by overfishing (Harrison and Stiassny 1999). Another example of the combined effects of inland fisheries and external pressures is the introduction of Nile perch (*Lates niloticus* (L.), Percidae) in Lake Victoria (Africa) and its supposed contribution to the declining numbers and possible extinction of many cichlid species in the lake. Several publications (see Harrison and Stiassny 1999 and references therein) stated that the combination of the explosion of Nile perch populations in the 1980s, fishing pressure (cichlids are caught as bait for the Nile perch fishery and by-catch), increasing lakeside agriculture, urbanization, and deforestation resulted in the decline of the cichlids and chronic eutrophication and anoxia in parts of the lake. Although in fresh waters actual extinction of species by fishing alone has not been demonstrated to date (Welcomme 1992; Marschall and Crowder 1996; Harrison and Stiassny 1999), the diverse, synergized interactions of fisheries practices, fisheries management measures and external forces determine the necessity to reduce, in addition to nonfishing impacts, every potential negative impact of inland fisheries on aquatic ecosystems.

Threats and constraints

Major impacts and threats on inland fisheries have already been mentioned throughout this paper (e.g. see Introduction). They can briefly be summarized (compare also Fig. 5) as follows:

1 Degradation of the environment and loss of fishery habitat are the predominant concerns for the sustainability of inland fisheries (FAO 1999). It is obvious that an impoverished inland water reduces benefits of inland fisheries to individuals and society, although recreational fishing may always be possible until the ecosystem has finally been destroyed (compare 'life cycle of fisheries', Smith 1986). However, in some countries (e.g. UK) there may still be the possibility to fish in artificial, intensively stocked, purpose-built stillwater fisheries which recently have grown in numbers (e.g. North 2002).

2 The multipurpose nature and use patterns of inland waters have created a climate in developed countries in which fisheries are usually not considered of sufficiently high priority or value and thus suffer in the face of economically and socially higher priorities, such as agriculture, hydro-electric power production and flood prevention (Cowx 1998a; Cowx and Collares-Pereira 2002).

3 The greatest short-term problems for inland fisheries arise from conflicts between local user groups, both intrasectorally (recreational vs. commercial fisheries, interactions between recreational fishing groups) as well as cross-sectorally (e.g. water-based recreational activities, conservationists, animal welfare or animal rights lobby groupings) (O'Grady 1995; Cowx 1998a, 1999a, 2002c). The latter cause conflicts over human interactions with aquatic organisms that have the potential to hinder management efforts to provide benefits from fisheries (AFS 1999). Various antifishing protests have occurred in several countries around the world (e.g. LaChat 1996; AFS 1999), consequently stopping, for example, catch-and-release

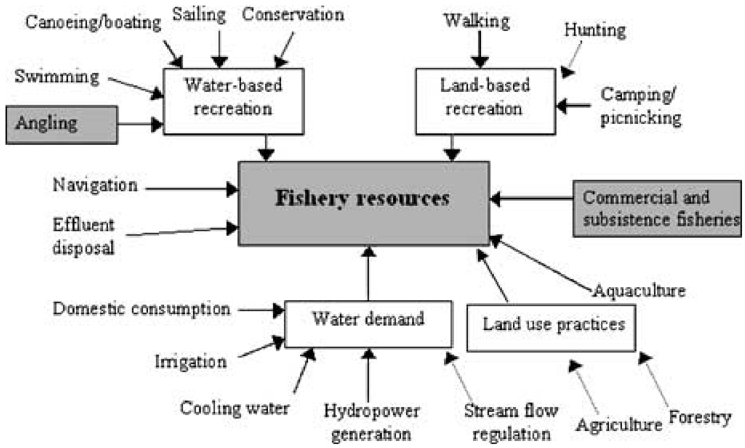

Figure 5 Stakeholders that typically affect fisheries and fishery resources in inland waters.

(Aas *et al.* 2002) and competitive fishing in some German federal states, and use of live bait fish in Norway, the Netherlands and Germany (Wortley 1995; Berg and Rösch 1998; Spitler 1998).

4 Low funding for fisheries research and management of inland waters is threatening the future of the entire inland fisheries sector (e.g. Shupp 1994; Post *et al.* 2002). Furthermore, high variable costs such as labour, reduced yield, contaminated fishes and changing consumer habits favouring certain fish species, *inter alia*, can endanger the cost-effectiveness and viability of commercial enterprises (e.g. Grosch *et al.* 2000).

5 As recreational fishing is a voluntary activity without the necessity to make profit, various constraints to angling have been elucidated in the scientific literature. For example, ageing of the human population and important changes in its economic, racial and ethnic composition have been suggested as forces leading to a decline in participation in freshwater fishing in North America (Murdock *et al.* 1992). Constraints are intrapersonal (psychological), interpersonal (social interactions) and structural factors (Fedler and Ditton 2000, 2001), and include a variety of variables that depend upon personal and situational characteristics such as age, gender, stage in life cycle, resource supply and type of recreational activity (Ritter *et al.* 1992). Perceived lack of time or money, lack of access to and knowledge of facilities, negative images of water quality and fish contamination, inconsistent delivery of satisfactory fishing products, and lack of a consistent positive image of fishing were identified as primary constraints to participation in USA (Fedler and Ditton 2000). Lack of time, low energy after work, child care responsibility, and the perception that fishing is boring have been found to be the most important constraints on recreational fishing expressed by the public in Norway (Aas 1995). In addition to lack of time (compare also Fedler and Ditton 2000, 2001), management itself (e.g. regulations and activities that do not result in acceptable catches of fish) is suggested to be a decisive constraint on angling (Ritter *et al.* 1992). Finally, the increased diversity of easily accessible, alternative leisure activities, and the proliferation of high-technology computerised leisure activities, has deterred the younger generations from participating in angling and outdoor pursuits (Cowx 2002c; Lyons *et al.* 2002).

Evolution of inland fisheries management and its paradigms in Europe

Fisheries management can be defined as the use of all types of information (ecological, economic, political, and sociocultural) in decision making that results in actions to achieve goals established for fish resources (Krueger and Decker 1999). The following section gives a brief historical overview of the evolution of inland fisheries management and its paradigms in Europe, and describes trends and current management practices.

Management history

Attempts at managing European inland fisheries are ancient (Welcomme 2001). Because mediaeval Europeans consumed great quantities of fish, fishing pressure increased in relation to population growth (e.g. Hoffmann 1996; Arlinghaus *et al.* 2002). By the

13th century, legislators already complained about overfishing (Hoffmann 1995). At the same time, important anthropogenic habitat modifications took place influencing the fishery resources negatively (Hoffmann 1995, 1996). As a result of declining fishery resources, European historians of mediaeval law recognized privatization of previously common or public fishing rights as a general phenomenon in the Middle Ages (Hoffman 1995). By about 1200, grants from kings or simple seizure put landowners in possession of all but the largest inland river fisheries (Hoffmann 1996). Between 1200 and 1400, development of markets and fishing rights took place. One-time lordly servants evolved into full-time fishers who paid annual monetary dues or, rarely, a share of the catch for the right to exploit the lord's water (Hoffmann 1995). Subsequently, mediaeval Europeans became aware of shifts in the availability and the limited nature of (renewable) fishery resources stimulating the evolution of fisheries management. Inland fishermen were sometimes grouped into guilds, which were charged with the exploitation and management of the resource (Welcomme 2001). Public authorities such as kings and lords undertook regulation of fisheries for both consumption and conservation motives (Hoffmann 1995, 1996). The first fisheries laws in Europe (13th century and thereafter) and other legislations aimed at preserving fish populations by controlling exploitation (Hoffmann 1995; Kirchhofer 2002). Thus, size of fish caught, gear and temporal restrictions and other regulations were put into force as far back as mediaeval times (e.g. Hoffmann 1996).

Evolving management paradigms

This simple fisheries management worked more or less until the Industrial Revolution in the late 1800s allowed rapid expansion of exploitation for an ever-growing market. At the end of the 19th century, as a result of environmental degradation, pollution (especially in rivers) and overfishing, inland fisheries of now industrialized countries were experiencing their worst conditions in history (e.g. Nielsen 1999; Arlinghaus *et al.* 2002). Consequently, the management paradigm of Maximum Sustainable Yield (MSY) began to spread during the 20th century as a result of a public awareness of the need to conserve natural resources (also called conservation paradigm) (e.g. Charles 1994; Caddy 1999; Nielsen 1999; Mace 2001). Since then, under a single-stock management paradigm, much scientific work focused on understanding and modelling population dynamics of exploited fish stocks (e.g. Schaefer 1954; Beverton and Holt 1957; Ricker 1958; Hilborn and Walters 1992), and the decades from 1900 to 1950 might well be called 'the golden age of fisheries management', particularly in the marine environment (Nielsen 1999). Since 1950, focus on MSY, as an objective to achieve the highest physical yield, has been challenged repeatedly (e.g. Larkin 1977), and possible objectives for management have been continually expanded (Nielsen 1999). For commercial fisheries, the management paradigm of Maximum Economic Yield (MEY) emerged because of the need to maximize profit and not yield *per se*, and thus manage fish populations efficiently (rationalization paradigm) (e.g. Charles 1994; Nielsen 1999). Clark (1973), however, demonstrated that in fisheries an orientation based on MEY could also lead to over-exploitation. MSY was also being challenged as an appropriate management objective for recreational fisheries because anglers value many components other than harvest (e.g. McFadden 1969; Moeller and Engelken 1972; Fedler and Ditton 1994). Moreover, because fisheries are components of the productivity of aquatic ecosystems, ecologists in the 1970s supplied the notion that the management of single species must be replaced by multiple-species management (Nielsen 1999). The accretion of additional concerns – economic, sociological, and ecological – into management of inland fisheries displaced MSY in the 1970s (Larkin 1977) to be replaced by a new paradigm of Optimum Sustainable (or Social) Yield (OSY) (Hudgins and Malvestuto 1996; Malvestuto and Hudgins 1996; Brown 1998). OSY was formalized by Roedel (1975) who assessed management from a variety of viewpoints. The basic tenets of OSY are that the appropriate goal for fisheries management includes a broad range of considerations (not just maximizing physical yield) and that a unique management goal exists for each fishery (Nielsen 1999). OSY thus greatly complicates inland fisheries management. Defining the elusive goal of OSY for a fishery is much more difficult than defining the MSY because fishery-specific information is needed about biological, ecological, economic and sociological aspects of fishery use (Nielsen 1999). The incorporation of the human dimension into OSY, especially for recreational fisheries (Malvestuto and Hudgins 1996), has also been named the social or community paradigm for (marine) fisheries (Charles 1994). With this background, it is surprising that most of the current success stories of inland fisheries management in the literature refer to single-species fisheries (e.g. Ebener 1997; Knight 1997), and yield prediction and the traditional concepts of MSY and MEY still dominate in the mind of politicians and fisheries managers in some European countries, such as Germany (e.g. Knösche 1998).

A new role for MSY and yield prediction?

With respect to recreational fisheries management, yield prediction seems to be inappropriate because individual anglers act as selfish benefit maximizers, not selfish profit maximizers (Table 3). However, if conflicts about allocation of fisheries resources arise intrasectorally (e.g. recreational vs. commercial fisheries), prediction of yield could be helpful in leading to socially acceptable resource allocation. For example, economically threatened commercial fisheries could be favoured by public authorities when access to the most productive systems has to be decided, because value and democracy would advantage recreational fisheries in allocation conflicts (Radonski 1995). Indeed this is the case for many salmon fisheries where recreational proprietorship is buying out the commercial ventures to secure exclusivity and maximize revenue by protecting the resources from overfishing by the commercial sector (Windsor and Hutchinson 1994). A family of empirical models exists to estimate yield in general or MSY more specifically derived from the generalized formula:

$$\mathrm{MSY} = x\mathrm{MB}_0$$

where B_0 is an estimate of the virgin unexploited biomass, M is the instantaneous natural mortality rate, and x takes the value of 0.1–0.5 depending on the stock characteristics. The foundations of the model descend from surplus production models which are holistic models that encapsulate the net effects of recruitment, growth and mortality in terms of biomass (see Hilborn and Walters 1992 for details). Because of the simplicity of surplus production models, the data requirements to parameterize them are less demanding (Welcomme 2001). Although approximate, such estimators provide a useful indication of the magnitude of a potential resource for planning and development purposes (Welcomme 2001).

Table 3 Selected differences between marine and inland fisheries of developed countries. Arguments subjectively considered as disadvantage for sustainable inland fisheries management are printed in italics.

Marine fisheries	Inland fisheries
• Open access	• Restricted access
• Less defined property rights	• Well-defined (group) property rights
• Global scale structures	• Small-scale structures
• Tragedy of the commons	• Less tragedy of the commons
• Free-riding behaviour more likely	• Less free-riding behaviour
• Reciprocal cooperation less likely	• Reciprocal cooperation more likely
• Commercial fisheries predominant	• Recreational fisheries predominant
• Selfish profit maximizers	• Selfish benefit maximizers
• Predominantly economic benefits	• Diverse use and nonuse benefits
• Top-down-driven traditional fisheries management, less involved fishermen	• Bottom-up-driven traditional fisheries management, fishermen more involved
• Rather reactive management systems	• Rather proactive (manipulating) management systems
• Interdependence between countries	• Less interdependencies between countries
• Massive damage of gears to habitat	• Less damage of gear to habitat
• Unequivocal overfishing tendencies	• Less overfishing tendencies
• *Less physical and ecological ecosystem diversity*	• *Great physical and ecological ecosystem diversity*
• *Low number of water bodies*	• *High number of water bodies*
• *Interconnected research and management units*	• *Independent research and management units*
• *A lot of marine fisheries research*	• *Less inland fisheries research*
• *Stocking practices not dominant management measure*	• *Stocking practices predominant management measure*
• *Long-term influence of human impacts*	• *Short-term influence of human impacts*
• *Fishermen dominant user group*	• *Diverse user groups, inland fishermen less dominant*
• *Marine fisheries important in economic terms*	• *Inland fisheries less important in economic terms*
• *High social priority*	• *Low social priority*

Empirical relationships for catches in rivers, which are also useful for European conditions, were summarized by Welcomme (2001). Various authors provided empirical regression models for yield prediction in lakes (see Leach *et al*. 1987; Kerr and Ryder 1988; Downing and Plante 1993; Scarborough and Peters 1993; Knösche and Barthelmes 1998; Ney 1999; Nissanka *et al*. 2000; Brämick 2002 for reviews and further references). Selection of a model to project fishery productivity in a particular system should be based on: (i) feasibility of data measurement; (ii) predictive power and (iii) similarity of the ecosystem to those in the regression data set (Ney 1999; Welcomme 2001).

However, with respect to MSY, the development of the precautionary approach in fisheries management has resulted in the fishing mortality associated with MSY (F_{MSY}) being thought of as a limit to be avoided rather than a target that can routinely be exceeded (Richards and Maguire 1998; Mace 2001). This has been named 'a new role for MSY' in marine fisheries management. However, in inland fisheries management, the abundant data requirements and the ignorance of economic and social considerations are likely to limit the usefulness of the 'new' MSY concept and F_{MSY} as a limit reference point. Furthermore, a sustainable inland fisheries management approach should abandon ill-equipped, ill-defined and mostly ineffective management paradigms (such as MSY). It seems dangerous to go on with MSY because misinterpretation by politicians and (traditional) fisheries managers is likely to occur. Instead, there should be a focus on OSY and EBM. The latter concept, however, whilst being academically appropriate may be premature (Grumbine 1997) because a precise definition of the desired target configuration of a rebuilt (Pitcher and Pauly 1998) or healthy ecosystem (Callicot and Mumford 1997) is difficult to quantify (Mace 2001). Before it can be accepted as a management tool for inland waters, EBM needs to be more advanced in terms of evaluating alternative ecosystem states, defining operational ecosystems objectives and specifying ecosystem management standards and performance measures analogous to those that currently exist for single-species management of fisheries (e.g. Larkin 1996; Mace 2001). The greater information requirements (e.g. more data for a variety of species) of EBM probably limits the future performance of ecosystem-wide approaches. This is particularly true in fresh waters because of the financial and human resource constraints imposed on the scientific community (Buckworth 1998; Mace 2001; Table 3). This calls for modifications to traditional inland fisheries management practices as prerequisites for sustainability in inland fisheries (Fig. 3).

Inland fisheries versus marine fisheries management

Currently, in Europe most inland water areas are privately owned and linked to land ownership. Many larger water bodies, however, are still publicly owned. Private and public owners of water bodies such as state, province, federal states or industries, lease fishing rights to commercial or recreational fishermen, associations, cooperatives, clubs or organizations, which are often charged (sometimes by law) with the duty to manage fisheries resources and the ecosystems (e.g. Bninska 2000; Salmi and Muje 2001; Fig. 5). Because of the duty of the inland fishing rights holder to manage the aquatic ecosystems properly and take into account the interests of nature conservation (e.g. the Netherlands, Raat 1990), fishery resources in Europe resemble 'quasi common pool' resources within private ownership. Well-defined property rights are, at the same time, basic conditions for sound and effective fisheries management because the 'tragedy of the commons' (Hardin 1968) concept is less likely to occur in inland fisheries than in the marine environment (see Ostrom *et al*. 1999 for details) where open access to fisheries resources predominates (Table 3). Generally, the small scale structures in inland fisheries, and, amongst other things (Table 3), the predominance of benefit and not profit-maximizing individuals (anglers), constitute excellent conditions for the future development of sustainable inland fisheries systems. However, because of the diverse structure of ecosystems and local conditions in inland waters, there is an urgent need to develop distinct local management systems. The arbitrary transfer of efficient systems from one place in Europe to another seems impossible. Equally, because of the marked differences between marine and inland fisheries (compare Table 3), transfer of marine management systems into fresh water also seems, in almost the same manner, impossible. Consequently, specific inland fisheries management systems have to be developed locally.

General trend

Welcomme (1997, 2000, 2001) and Welcomme and Bartley (1998) described two main strategies for the management of inland waters depending on different societal views of natural resources (Table 1). Conservation approaches tend to be dominant in the more developed economies of temperate latitudes where food surpluses allow for alternative uses of natural resources. In less developed countries, food shortages are such that a more production-orientated approach is needed. Thus, in developed countries, management is orientated towards satisfying the recreational fishing community, although in recent years an increasingly protectionist lobby enjoys growing influence (see Welcomme 2001 for more details). In industrialized countries, there have been long-term trends to suppress commercial, in favour of recreational, fisheries in natural waters whilst using intensive aquaculture to produce selected freshwater species for profit (Welcomme 1997). In future, recreational fisheries will certainly gain further importance in developed countries and dominate the inland fisheries system (e.g. Smith 1986; Pikitch *et al.* 1997; Welcomme 1997, 2001; Arlinghaus *et al.* 2001, 2002). Moreover, recreational fishing is also becoming popular in tropical countries (Mace 1997; Cowx 2002c).

Regardless of this general trend, inland fisheries management systems vary throughout Europe (e.g. O'Grady 1995). However, central to all of them are: (i) a hierarchical decision-making structure and (ii) the predominance of stocking and diverse regulations coupled with habitat management practices as traditional management measures (Cowx 2000, 2002a, b,c).

Traditional management hierarchy

The hierarchy of traditional inland fisheries management decision making in Europe can be broadly divided into a 'top down' component which is formed by public decision making and enforcement at higher levels (regional scale), and a 'bottom up' component which is formed by (private) decision making and enforcement in lower levels (local scale) (Fig. 3).

In higher levels, the responsibilities for inland fisheries management are shared among governmental public authorities (e.g. the Environmental Agency (EA) in England and Wales, or Fishery Boards in some German Laender), which operate within the national legal framework. Generally, the entire national legislative framework concerning water resources, environment, nature, fisheries or animals (e.g. Animal Protection Law, Water Resources Act, Nature Conservation Law) affects the inland fisheries management sector (e.g. Hickley *et al.* 1995; Rasmussen and Geertz-Hansen 2001). Public fishery authorities are often politically too weak to take effective actions to deal with policy problems in inland fisheries management (Preikshot 1998). Furthermore, in Europe there is often no structured communication between water institutions and fisheries institutions, which sometimes gives rise to conflicting situations (Raat 1990; Welcomme 2001).

In some countries, inland fisheries are dealt with by specific fisheries legislation. There are examples of nationwide fisheries laws (e.g. France, Poland; Bninska 2000; Welcomme 2001) or diverse fisheries laws in federal systems (e.g. 16 fisheries laws in Germany). In many other countries, however, there is no separate legislative treatment of inland fisheries (Welcomme 2001). Within the legal framework of the country or federal states, public authorities responsible for fisheries issues, inland fisheries organizations, cooperatives, associations, clubs, syndicates or even individual fishermen, enforce regulations and manage the waters on which they have fishing rights (Fig. 3, e.g. Raat 1990; Hickley *et al.* 1995). This is practical inland fisheries management at the local scale and lower operational levels which, in some countries, is supported by NGOs, regional advisory committees, research institutes, local public fisheries authorities or governmental agencies such as the EA in England and Wales (Lyons *et al.* 1999; Hickley and Aprahamian 2000). In some countries, institutions at the intermediate decision-making level have been created, such as fisheries regions in Finland (Sipponen 1998, 1999; Salmi *et al.* 2000; Salmi and Muje 2001). Furthermore, at the lower level, there are sometimes voluntary codes or informal institutions (Ostrom *et al.* 1999) adopted by individuals, e.g. an angling club (Hickley *et al.* 1995). In selected European countries, higher levels of the decision-making tree, such as the EA, take the lead in inland fisheries management, provide an integrated fisheries service, develop management plans and link institutions responsible for water management with those responsible for fisheries management (Lyons *et al.* 1999; Welcomme 2001). In other countries of Europe, such as in Germany and the Netherlands, the local inland fisheries management remains more or less in the hands of inland fishermen or angling clubs and associations, which are often not well trained in fisheries science or ecology (compare also Templeton and Churchward 1990 in UK). Besides, honorary and unsalaried fisheries managers of angling organizations and clubs (e.g. Walder and Van der Spiegel 1990) are often single-species orientated, driven by the interests and pressure of club members

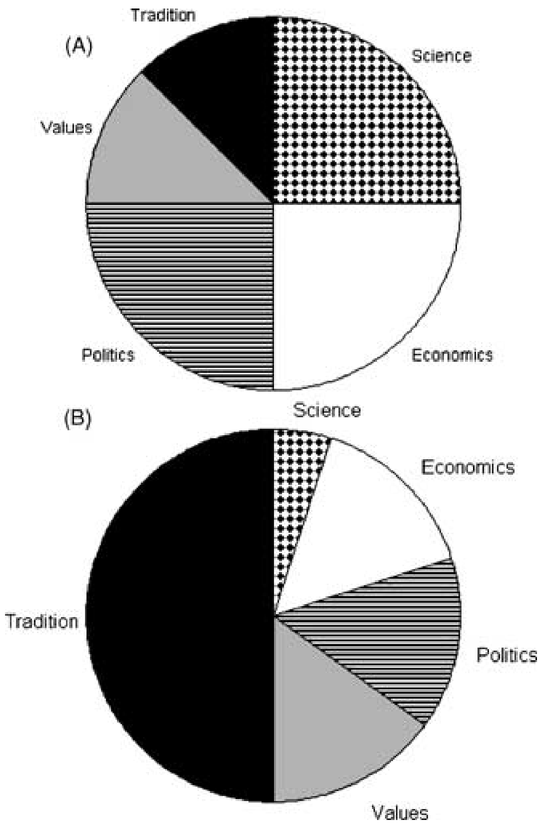

Figure 6 Pie diagram of some major factors involved in decision making and (marine) fisheries (top, from Policansky 1998) and recreational fisheries management of developed countries (bottom). The relative sizes of the segments are only rough approximations, but the science segment rarely is, nor should it be, much larger than depicted on the top (25%).

and sometimes overtaxed with the task to manage properly complex ecosystems and food webs under conditions of uncertainty (e.g. Ludwig *et al.* 1993; Francis and Shotton 1997). This means that, for example, in Germany there are management practices which are based more on tradition or trial and error than on sound scientific or ecological advice (e.g. Klein 1996; Fig. 6), and an integrated approach taking into account all stakeholders is seldom in practice (see also Raat 1990, the Netherlands; Smith and Pollard 1996, Australia). Consequently, inland fisheries management in countries such as Germany is a dispersed activity, and integrated sustainable inland fisheries management is far from prevalent (Arlinghaus *et al.* 2001). Fuelling the problem, volunteer fishery managers of angling organizations often do not accept that some of their traditional management practices are unsustainable, partly because of their poor ecological or fisheries science background. In addition, people usually prefer immediate results over larger rewards in the (uncertain) future (Fehr 2002) and given the complexity of aquatic ecosystems, it is often difficult for fisheries stakeholders and managers to see and understand the larger picture of the (eco- or fishery) system as a whole (Ascher 2001). In practice, most people are overtaxed to make reasonable decisions given complex, uncertain and slow (time lag between cause and effect) processes of dynamic aquatic ecosystems (Dörner 1996). Thus, they tend to focus on local conditions and short-term solutions, and extrapolate from these experiences to the larger frame. Furthermore, inland fishermen may suffer from the same shifting baseline syndrome described for fisheries scientists (compare Lappalainen and Pönni 2000 for eutrophication perception of Finnish anglers, Post *et al.* 2002). This syndrome suggests that each generation accepts as a baseline the fish stock size and species composition that occurred at the beginning of their careers or lives, and uses this to evaluate changes (Pauly 1995). Against this backdrop, it is often difficult to shift to an alternative state that attempts to improve the habitat conditions of the water body towards its perceived original state because this would probably result in a decline in the fishery.

Dominant traditional inland fisheries management measures

Traditional inland fisheries management is carried out on three levels: the fishery, the fish and the aquatic ecosystem (Fig. 7; Welcomme 2000; Cowx 2002b,c). By far the most dominant traditional inland fisheries management measures in Europe are regulations (targeting the fishery) and stocking practices (targeting the fish stocks) (see case-studies in Van Densen *et al.* 1990; Cowx 1998b, 2002a; Müller and Bia 1998). To a lesser extent, inland fisheries managers use habitat management techniques to improve access to, and quality of, fishing, improve degraded habitats and increase the production potential of the fishery (targeting the ecosystem) (see Cowx and Welcomme 1998; Welcomme 2001 for details).

Targeting the inland fishery: regulatory techniques

Regulations are the most ancient inland fisheries management measures (Hoffmann 1995, 1996). Today, most regulations in inland fisheries management are promulgated in laws and byelaws and are

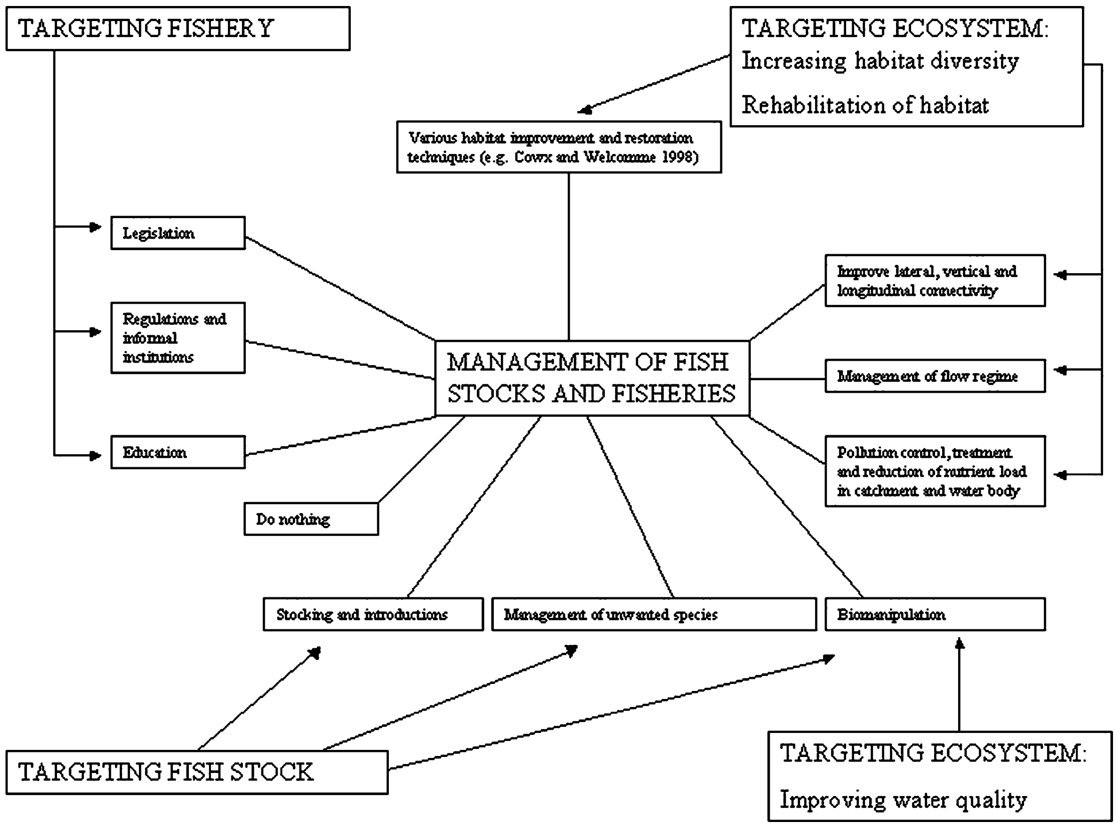

Figure 7 Interventions used in the management of fish stocks and inland fisheries. Normally, appropriate management programmes will require the integration of several techniques (Cowx 1994a; modified).

sometimes further extended by the holder of the fishing rights (e.g. commercial fishermen or angling club rules) or by informal institutions on a voluntary basis (Hickley *et al.* 1995). The purposes of inland fisheries regulations include managing social issues (e.g. attempt to distribute harvest more equitably), preventing overfishing, maintaining a suitable stock structure, and manipulating an aquatic community (e.g. predator–prey interactions). Regulations include technical measures (e.g. closed area, closed season, gear restrictions, catch reports) as well as input (access and effort control, e.g. examinations, licences) and output (e.g. catch limits, size limit, catch-and-release) controls (Table 4; see Hickley *et al.* 1995; Noble and Jones 1999; Welcomme 2001; Cowx 2002c for details). A lot of regulations such as catch limits or size limits are predominately directed towards selected commercially valuable (commercial fishermen) or highly appreciated (anglers) species of the fish community. However, most regulations were not determined by scientific advice but rather set arbitrarily through consensus with little biological justification (Radomski *et al.* 2001). Irrespective of this, enforcement of regulations is difficult and expensive (e.g. Hemming and Pierce 1997), but because pressure on habitat and fish stocks will continue to intensify, there can be little doubt that the role of regulations in inland fisheries management will increase in future (Noble and Jones 1999). However, formal scientific experiments with adequate replication and control (Hurlbert 1984; McAllister and Peterman 1992) are needed to evaluate fully the (ecological) effects of regulations in fresh waters (Nordwall *et al.* 2000), and to reduce the degree of arbitrariness in inland fisheries regulations (Policansky 1993; Johnson and Martinez 1995; Wilde 1997; Radomski *et al.* 2001). Irrespective, noncompliance of fishers and anglers with regulations and illegal harvest may reduce the efficiency of even the best planned fishery regulation (Gigliotti and Taylor 1990; Pierce and Tomcko 1998).

Targeting the fish stock: stocking and introductions

Stocking and introductions are ancient inland fisheries management practices and the prevalence of

Table 4 Techniques for regulating inland fisheries and the ecological requirements of the fish stocks which are addressed.

Regulatory technique		Numerical population size	Broodstock protection	Undisturbed spawning	Free passage	Fish welfare
Technical measures						
	Closed areas	*	*		*	
	Close season		*	*	*	*
	Type of gear					*
	Catch reports	*				
Input controls						
	Licenses	*	*			
	State-regulated access	*	*			
	Ownership	*	*			
Output controls						
	Catch limits	*				
	Catch-and-release	*	*			*
	Size limit[a]	*	*			*
	Sale of fish	*	*			*

[a]Size limit also addresses length distribution of the population. Symbol (*) indicates the ecological requirements that the different regulatory techniques target on.

common carp culture since the Middle Ages (Balon 1995) is only one example of stocking and introduction measures which led to the transfer of fish species throughout the world (Welcomme 1991; Cowx 1996, 1998b). The development of artificial propogation techniques and the spread of hatcheries and aquaculture facilities since the late 1800s accelerated the number of stocking programmes (Heidinger 1999). Today, many thousands of stocking events, involving millions of individual fish, take place annually in so-called managed or culture-based fisheries throughout the world (Hickley 1994; Petr 1998).

Stocking, introduction and transfer of fish are considered valuable management tools which complement the physical rehabilitation of the environment (Cowx and Welcomme 1998) and they are frequently used by fisheries owners and managers in the belief that they will improve the quantity and quality of catches and have long-term beneficial effects on fish stocks (Cowx 1994b). Inland fisheries managers intensively stock and introduce fish for four main reasons (Fig. 8; Cowx 1994b; Cowx and Welcomme 1998): (i) mitigation or compensation (stocking with native species to compensate for a disturbance caused by human activities, e.g. lack of spawning habitats, lack of connectivity due to damming); (ii) maintenance (stocking to compensate for recruitment overfishing); (iii) enhancement (stocking to maintain fisheries productivity at the highest possible level, combines stocking for compensation and maintenance and seeks to direct the off-take from the waters towards a certain number of valuable or attractive species) (iv) conservation (stocking to retain stocks of a species threatened with extinction).

Introduction of new species takes place to: (i) establish new fisheries; (ii) fill a vacant niche; (iii) control pests; (iv) control water quality; (v) develop aquaculture and (vi) fulfil aesthetic and other reasons (see Welcomme 2001 for details and decision tree). A total of at least 2673 introductions of 291 species into 148 countries have been recorded worldwide (Welcomme 1998a; compare also FAO Database on Introduction of Aquatic Species DIAS, http://www.fao.org/fi/statist/fisoft/dias/index.htm, latest access 18 July 2002). In Europe, at least 113 species of fish and eight species of large crustacean have been introduced in 566 recorded introductions (Welcomme 1991).

Despite widespread application of stocking and introduction programmes in inland fisheries management, relatively few programmes have been properly evaluated (e.g. Steel *et al.* 1998), and the evidence suggests that stocking exercises rarely led to any long-term tangible benefit (Cowx 1994b). This appears to be the result of indiscriminate stocking, without well-defined objectives or *a priori* appraisal of the likelihood of success of the exercise (Cowx 1994b). In some cases (e.g. hatchery-reared salmonids in Austria), stocking is even obligatory wherever fishing licenses are sold (Weiss *et al.* 2002). In some European countries, such as Germany, the holder of

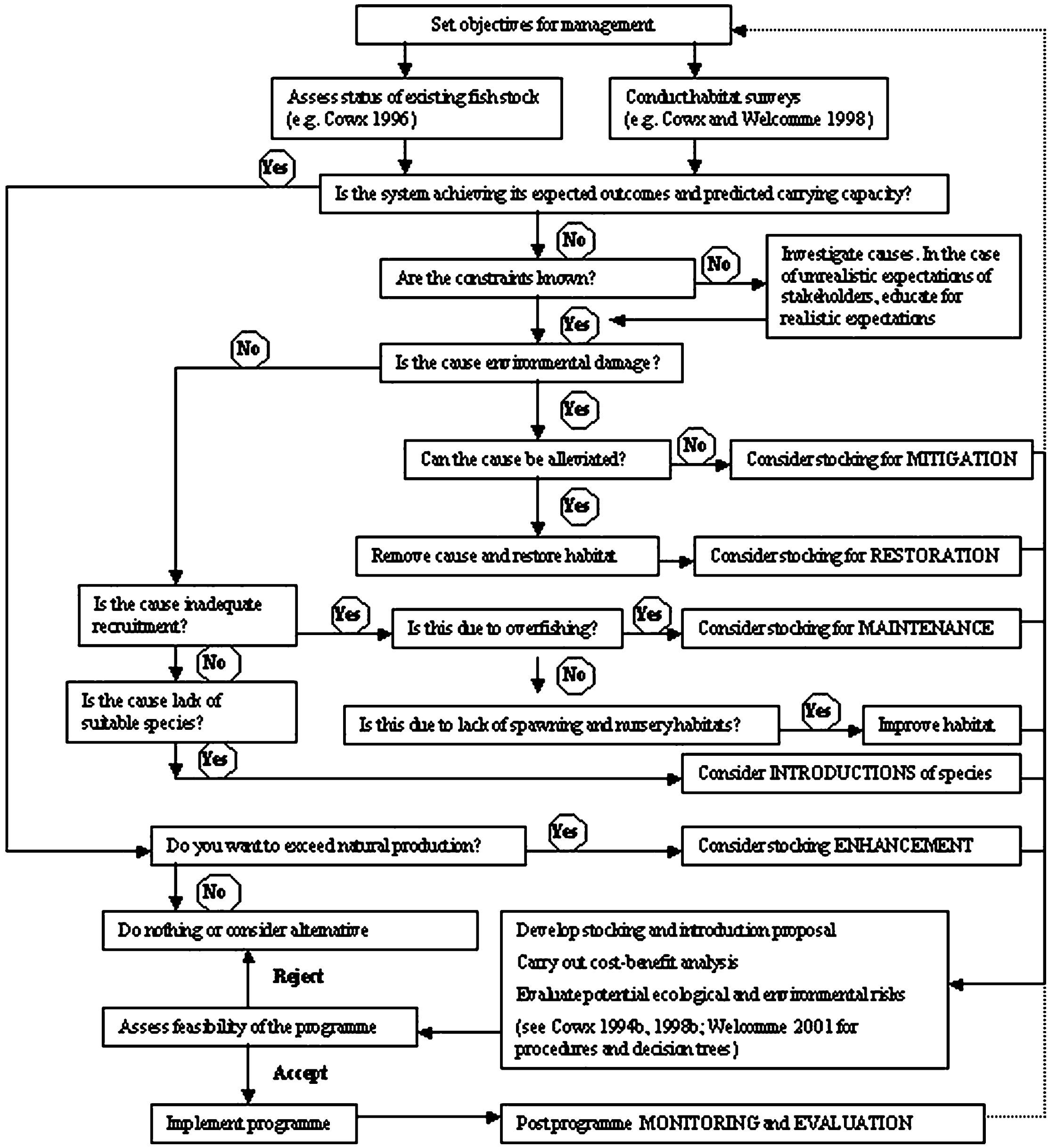

Figure 8 Suggested strategy for planning various types of stocking and introduction exercises to minimize the potential risk, maximize the potential benefit and evaluate the success of the project in inland fisheries management (after Cowx 1994b; Welcomme 2001, modified).

the fishing rights (and in reality nearly everyone due to lack in enforcement) is able to buy stocking material from private aquaculture enterprises and stock aquatic ecosystems. The choice of stocking regime is often based on a best guess, determined by external constraints (e.g. size and numbers of seed available) (Welcomme 2001), or driven by 'insider relationships' with stocking material traders. The stocked species are often not determined from an ecological point of view but are those species that are most valuable or attractive to sell or to catch, and thus demanded, e.g. by anglers. One extreme, but not uncommon, view of local fisheries managers (e.g. of angling clubs) seems to be 'that the best way to manage a freshwater fishery is to put in the fish the anglers want and ensure that most are caught' (Elliott 1995). Stocking often seems to be a prophylactic traditional inland fisheries management practice undertaken by habit (Klein

1996). Such stocking practices have been questioned by fish ecologists and environmental lobby groups (e.g. Freyhof 2002). Stocking practices, as explained above, are not in agreement with the principles of sustainable inland fisheries management such as the precautionary approach (see Section 2, FAO 1996) which, inter alia, call for actions to be reversible (Buckworth 1998). Furthermore, especially with respect to recreational fisheries in many European countries, the precautionary approach implies that very conservative management measures are required because of the paucity of scientific information available to underpin advice (Richards and Maguire 1998). Thus, there is a need for fisheries managers to be more aware of the possible negative impacts of stocking (Section 3; Cowx 1994b, 1996, 1998b); and a thorough planning of stocking and introduction programmes is essential (Fig. 8; Coates 1998; Cowx 1998b; Welcomme 2001) because of the potential risks associated with fish movements (Pearsons and Hopley 1999; Ham and Pearsons 2001). As a result, in some countries, e.g. Denmark, stocking can only be undertaken with the prior permission of the public authorities (Rasmussen and Geertz-Hansen 2001).

There are, however, exceptions to the arguments against stockings. In some cases, the justification for stockings is acceptable, for example, to compensate for loss due to environmental interventions such as pollution, river engineering or a man-made obstruction to migration such as a dam, or to create a put-and-take fishery in an enclosed water body. Furthermore, where fish species are on the verge of extinctions or where bottlenecks to natural recruitment cannot be eliminated, continuous stocking seems appropriate (Cowx 2002a). This is particularly relevant where added benefits can accrue from the stocking programme, e.g. the River Thames Salmon Restoration Scheme which has considerable environmental spin offs, particularly the public perception that the river is now clean and the increased responsibility of water users to ensure this status is maintained (Mills 1990). Another example encompasses highly modified or artificial, small purpose-build water bodies where circumvention of recruitment failure is almost impossible. Under such circumstances stocking may lead to high angler benefits and reduce the fishing pressure on more natural fisheries with self-reproducing populations where empirical evidence suggests that stocking may be superfluous (Salojärvi and Ekholm 1990; Salojärvi and Mutenia 1994; Welcomme 2001). However, intensive stocking should be limited to closed water bodies, e.g. small stillwater fisheries where the probability of escape of stocked fish is low and the likelihood of successful stocking is higher (e.g. Moehl and Davies 1993; Steel *et al*. 1998). Furthermore, it is advisable to ensure that the broodstock from which the stocking material is derived is drawn from native populations of the catchment (Welcomme 1998a).

With respect to successful stocking programmes, the few available empirical data sets showed that there appears to be an interplay between area of stocked system and stocking rates. They indicate that yield is often positively related to stocking rate (Welcomme 2001). However, because of compensatory processes (Salojärvi and Mutenia 1994; Lorenzen 1996a) there is a levelling off or decline of yields at high fish densities. Furthermore, yield per unit area is often inversely related to the area of stocked system (Welcomme 2001). Therefore, stocking has generally proved more effective in smaller water bodies. However, this supposition cannot be readily validated because there is a tendency to stock fish at lower densities into larger water bodies (Welcomme 2001).

Generally, the key process governing the outcomes of stocking in the absence of natural reproduction is density dependence on body growth and size dependence on mortality (Lorenzen 1995, 2000). In combination, these processes result in density-dependent mortality (Lorenzen 2000). Under conditions of low natural reproduction and low density-dependent mortality after stocking, there are at least three management guidelines for stocked fisheries governed by density-dependent growth and size-dependent mortality (see Lorenzen 1995; Welcomme 2001 for details). First, the optimal stocking regime is dependent on the harvesting regime and vice versa. Second, potential production from stocked fisheries is inversely related to size at which fish are harvested. Third, concerning length of stocked fish, for a similar level of productivity, the numbers that need to be stocked decrease in a nonlinear way as size increases. This is the consequence of the allometric mortality–size relationship (Lorenzen 1995, 1996b, 2000). The biomass of seed that needs to be stocked to achieve a given level of yield increases with increasing seed size, and so does the cost of producing the individual fish. The issue of cost-effectiveness is an important component of efficient stocking programmes (Welcomme 1998a) and deserves more attention (Langton and Wilson 1998).

Several studies have elucidated the dynamics of stocked fisheries in greater detail and led to the development of practical assessment tools and general insights for analysing and understanding stocked systems (see Botsford and Hobbs 1984; Cuenco 1994; Lorenzen 1995, 2000; Welcomme 2001 for details and stocking equations). An example of a mechanistic (i.e. incorporating key mechanisms such as growth and mortality) stock assessment model is given by Lorenzen *et al.* (1997), while Lorenzen *et al.* (1998) provide an example of an empirical (i.e. describing observed relationships between stocking density and yield by regression equations) stocking model.

Irrespective, there is another issue of stocking which needs careful consideration. In addition to responses of fish populations to fishing and enhancement (e.g. stocking, production side), there is the response of fishers and anglers to changes in the abundance of fish (consumption side). Several studies showed that fishing effort and harvest may increase linearly with stocking rates as an index of fish abundance without an improvement in quality of fishing as measured by catch per effort (e.g. Shaner *et al.* 1996; Cox and Walters 2002a). Quantifying numerical and functional responses for recreational fisheries remains elusive (Johnson and Carpenter 1994) because they appear to depend on complex angler behaviours and decisions (Post *et al.* 2002). However, intensive stocking may result in a rapid angler response, raised angler expectations and finally higher exploitation level which may outpace the fisheries managers best stocking efforts (coined 'paradox of enhancement', Johnson and Staggs 1992).

Targeting the ecosystem: habitat management

Habitat management aims at rehabilitation of ecosystems and encompasses increasing fish habitat diversity and improving water quality (see Cowx and Welcomme 1998; Roni *et al.* 2002 for reviews). Habitat techniques used in inland fisheries management range from simple measures such as bankside vegetation cutting (e.g. Templeton 1995) or creation of artificial gravel beds (e.g. Swales 1989) to complex tasks such as the restitution of longitudinal, lateral and vertical connectivity in regulated rivers (Cowx and Welcomme 1998; Orth and White 1999; Roni *et al.* 2002). Habitat management is a tool which deserves considerably more attention in future inland fisheries management because it should be a long-term (although often expensive, e.g. Weiner 1998; Sheehan and Rasmussen 1999) solution to improve fish stocks and the quality of inland fisheries (e.g. Cowx 1994a,b; Lehtonen 1999; Williams *et al.* 1999). However, effective habitat management, such as environmental engineering or rehabilitation techniques, needs full consultation with water resource managers and environmental experts and an integrated approach (Cowx 1994a; Bradshaw 1996; Cowx and Welcomme 1998; Lehtonen 1999) because inland fisheries managers and fisheries authorities rarely have the political and financial power to implement complex habitat management measures alone (e.g. Ross and Loomis 1999; Knudsen and MacDonald 2000). In fact, many large scale management programmes to improve physical state of habitats are conducted by nonfishery players (e.g. nature conservation authorities) without consideration of interests of inland fisheries stakeholders. This, inter alia, has led to the focus on using stocking and regulations in traditional inland fisheries management.

In contrast to the body of evidence regarding the beneficial effect of habitat management (see Section 5), many case histories suggest that people (e.g. fisheries managers) tried to use technologies such as the production of excessive stocking material in hatcheries as substitutes for ecosystem functions (e.g. Meffe 1992). Management has often been based on the belief that natural ecological processes comprising a healthy ecosystem can, to a large degree, be replaced, circumvented, simplified, and controlled while production is maintained or even enhanced (Meffe 1992; Williams *et al.* 1999). However, experiences have shown that this is not true. In the Columbia River (USA), for example, despite the billions of dollars invested in technological solutions (first hatcheries and fish ladders, later screens at turbine intakes and irrigation diversions, then barging and trucking of juveniles fish around dams), salmonid populations continued to decline (Williams *et al.* 1999). If it is possible to remove or minimize the cause of the degradation of the ecosystem (and thus of the fishery), this course of action should be taken. The fishery may then recover without other traditional management practices, such as stocking, which are often not only ineffective but also expensive (and thus a waste of resources). Prioritization of the key cause(s) of damage and appropriate remedial action can have a dramatic positive effect on fish species diversity and productivity (see Cowx 1994a, 2000, 2002b; Collares-Pereira *et al.* 2002a and case-studies therin). It has to be kept in mind that provision of even the best mitigation measures (such as careful evaluated stocking programmes) can never fully compensate for losses of function or structure (Lucas and Marmulla 2000). Habitat improvement is the most desirable option because it should lead to long-term sustainable improvement with minimal deleterious ecological impact (Cowx 1994a,b, 2002a). Inland fisheries managers should be prepared to meet these ever-increasing challenges regardless of the type or quality of fishery they manage, be it in pristine waters or highly urbanized watersheds (Panek 1997). The main challenge for inland fisheries management may be named the 'habitat management first – stocking for mitigation and compensation last' paradigm (compare Langton *et al.* 1996; Williams 1997; Fluharty 2000). Altogether, for the future this calls for habitat management to be the primary inland fisheries and ecosystem management practice in industrialized countries.

Having said this, care is needed not to emphasize this strategy in every occasion. Restoration of aquatic habitats towards pristine conditions, which is the objective of most ecosystem managers, is a utopian view (Welcomme 1995; Wolter 2001). Many human-induced impacts on aquatic ecosystems are irreversible (e.g. heavy modification of river channels to control floods or provide navigation). More recently, demands for water resources, e.g. hydro-electricity, have created new impacts. Furthermore, many of the rehabilitation schemes are localized dealing with only small sections or reaches of the ecosystem (e.g. meso scale, Van Zyll de Jong *et al.* 2000) and ignore the problems associated with adjacent sections (Cowx and Collares-Pereira 2002). For example, water quality problems upstream invariably will have an impact downstream which will compromise any physical habitat improvement measures. Similarly, constructions of barriers (dams and weirs) in downstream reaches will impede upstream migration of fish and impose a serious bottleneck to recruitment, if, as it is often the case, they are not addressed. In addition, the cost of improvement schemes is high and financial resources necessary to undertake the job in a thorough manner are rarely available. Also, many of the improvement schemes conflict with the water resources activities in the catchment, especially hydropower generation and flood alleviation in river ecosystems. Consequently, most schemes are unable to provide all the features of the habitat predicted as necessary to ensure a return to the former status.

In addition, there is a naivety about the response of fisheries to habitat improvement measures because of lack of information from existing schemes. It must also be recognized that rehabilitation is not a reversal of the degradation because ecosystem dynamics are far more complex than merely reinstating the habitat for a particular species or fish community. This is conceptually illustrated in Fig. 9 (after Bradshaw 1996; Cowx 2002a) which suggests how communities respond to habitat manipulations either by: (i) increasing overall standing stock at the expense of biodiversity or (ii) increasing species diversity but at much lower standing stock than expected. The deviation from the expected restoration goal arises because of the loss or disruption of functional elements of the ecosystem during the degradation process that cannot be reinstated.

The issue can be summarized by the following statement (adapted from Welcomme 1995): Improvement of the situation demands stricter management of aquatic systems through protection of the few that remain in a relatively pristine state, and those modified should be rehabilitated if social, political

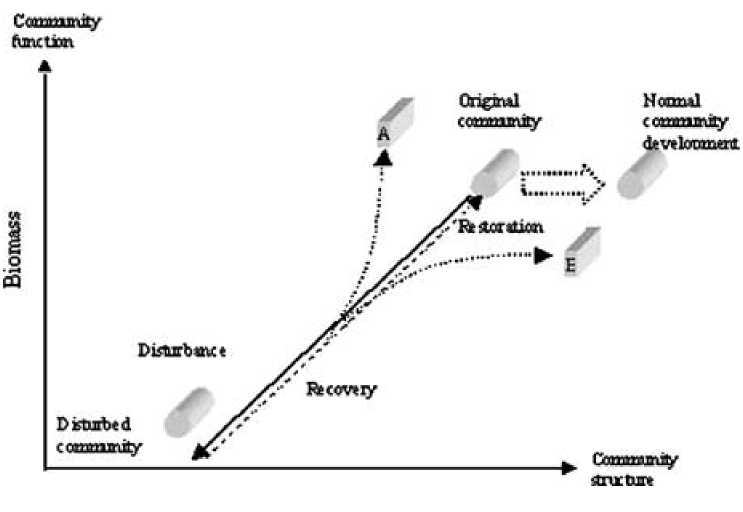

Figure 9 Schematic presentation of the process how rehabilitation of fish communities typically proceeds. As a result of habitat rehabilitation, fish populations often either increase overall standing stock at the expense of biodiversity (A) or increase species diversity but at much lower standing abundance (B).

and economic conditions allow. Should this not be possible, approaches for the mitigation (e.g. stocking) of externally imposed stress should be sought and applied.

Conclusion with respect to sustainability

To conclude, the situation in many inland fisheries of industrialized countries comprises: (i) nonexistence of integrated ecosystem management and precautionary approaches; (ii) widespread adoption of stocking and introduction practices without thorough planning and evaluation (which is contradictory to the precautionary approach); (iii) predominance of the management principle 'stocking rather than habitat management'; (iv) lack of adoption of sound scientific (fisheries or ecological) advice; (v) high degree of arbitrariness (e.g. regulations) and (vi) lack of a well-developed fisheries management framework and process (Fig. 10) to direct traditional inland fisheries management systems and associated practices towards the principles of sustainable management of inland waters (see Section 2). Thus, several management mechanisms and measures have to be altered or adapted for inland fisheries systems to approach sustainability (compare Section 2 and Fig. 3), which would, inter alia, also improve the dialogue between inland fisheries on the one hand and various stakeholders such as nature conservationists on the other.

Approaching sustainable inland fisheries – possibilities and constraints

The complex nature of the sustainability concept and the numerous factors that currently preclude sustainability in inland fisheries management (see above), highlight many mechanisms for altering current management practices. Of the approaches available, two major areas seem promising for the traditional inland fisheries management systems in Europe, and have led to several successful outcomes worldwide: (i) rehabilitation of habitat in running waters and (ii) a combination of nutrient reduction and biomanipulation in standing waters. The different ecological dynamics of flowing and standing aquatic systems and the divergent predominant constraints to (ecological) sustainability within flowing (mainly habitat modification) and standing waters (mainly 'cultural eutrophication') explain why

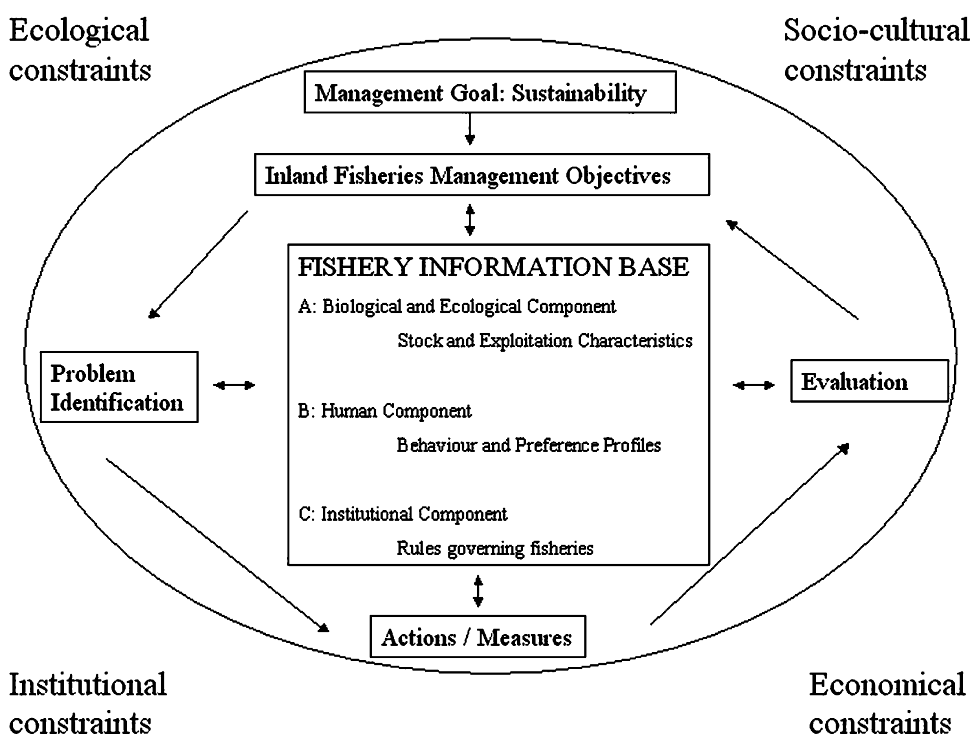

Figure 10 Framework for decision making in inland fisheries management (after Krueger and Decker 1999; modified). The steps of the management process are shown as they cycle around the fishery information base and are set within the management environment, which may constrain management practices.

promising integrated management approaches for each system are treated separately. It would have been preferable to develop recommendations that were more specific and more immediately applicable for inland fisheries management, but this would have required oversimplification and distortion of a very complex situation.

Running waters

Serious conflicts between various stakeholders in EBM and the disciplinary and fragmented structure of many water resources management systems have created the perception of many involved in the progress of inland waters management that expressing one's views and interests antagonistically is the only appropriate way forward (Preikshot 1998). Consequently, negotiation and policy making often become extremely difficult in natural resources management. Diverse players may, however, often have more in common than they realize (Preikshot 1998). This is particularly true for basin-wide river/stream management where anthropogenic habitat modification imposes the greatest threat for both inland fisheries and biodiversity (of freshwater fish) (e.g. Welcomme 1992, 1995, 1999; Schiemer *et al.* 2001). The deterioration of riverine habitats is reflected by the high number of endangered riverine and migratory fish species worldwide (e.g. IUCN 2000; Collares-Pereira *et al.* 2002a), which constitute not only natural capital with an intrinsic value providing important ecosystems services (Holmlund and Hammer 1999), but they are also commercially and recreationally valuable for inland fisheries. These relationships suggest that rebuilding of lotic habitats and natural flow regimes should be the common goal of the majority of stakeholders involved in natural resources and watershed management (e.g. environmental agencies, nature conservationists, NGOs, and sometimes society), including inland fishermen and their organizations and lobby groups. This task is difficult (e.g. Heede and Rinne 1990), needs strong cooperation between and across all stakeholders and public authorities, and thorough planning and evaluation (see for example Cowx 1994a, 2000), but it is not overwhelming (see below). Owing to societal priorities such as flood control, the changed property rights regime of the former floodplain which today is utilized by agriculture and habitation, and the deepening and narrowing of the river channels, rehabilitation of most of the engineered rivers and waterways is probably unrealistic (e.g. Welcomme 1995; Williams *et al.* 1999; Wolter 2001). However, in contrast to expensive and often arbitrarily adopted stocking practices, relatively 'simple' instream habitat improvement measures which modify flow and bottom substrate or provide direct cover (Table 5; Swales 1989; Cowx and Welcomme 1998; Hodgson and Eaton 2000) have often led to long-term beneficial effects on riverine fish communities (see review of Swales 1989 for numerous references and case-studies). In particular, highly degraded running water systems may benefit from simple habitat management. Wolter (2001), for example, suggested that in German waterways (including artificial systems such as navigation canals) the rehabilitation of natural shoreline structures from virtual nothing to 20% of the bank line should result in substantial improvement of fish diversity, and contribute to species conservation and persistence of viable populations of threatened riverine fishes without altering the waterways' primary navigation function. Furthermore, properly planned and adopted integrated watershed and river fisheries management approaches have demonstrated that carefully executed habitat improvement works can result in successful fisheries management and rehabilitation (e.g. Pajak 1992; Schmidt *et al.* 1997; Thorn *et al.* 1997; Cowx 2000). This indicates that when river engineering modifications have degraded river habitats, instream habitat management, land treatments and acquisition of riparian corridors are necessary to rehabilitate habitat and provide viable inland fisheries (Swales 1989; Thorn *et al.* 1997; Weiner 1998). Furthermore, in many flowing systems successful stocking and regulation management seem to be dependent on habitat quality (Thorn *et al.* 1997).

There are now many examples of successful habitat improvement activities in running waters (e.g. O'Grady and Duff 2000; Hughes *et al.* 2001; Jurvelius and Auvinen 2001; Souchon and Keith 2001). Much of these, however, target single-species stock recovery, especially for salmonids, although attention is now focusing on nonsalmonid fish populations in lowland rivers (Hodgson and Eaton 2000). However, habitat management that increases the stock of a target species may be a vehicle for other riverine fish to re-establish because suitable spawning substrate (e.g. gravel) and microhabitat (see Mann 1995) becomes available. Freyhof (2002), for example, called Atlantic salmon an important 'flagship' species because many improvements concerning water quality and aquatic habitats for salmon have improved conditions for other riverine species in

Table 5 Measures used in river habitat management and rehabilitation (derived from Cowx and Welcomme 1998).

- Local actions (instream habitat management)
  - Structures which impound and modify stream flow
    - Various measures to recreate pool-riffle characteristics
    - Woody debris obstructions
    - Low dams and weirs
    - Current deflectors
  - Structures which provide cover, refuge, food and stabilize banks where necessary
    - Artificial or natural cover devices, e.g. by trees, bushes, branches, platforms, boulders
    - Measures to enhance instream and riparian vegetation
    - Woody material to protect banks
  - Structures or treatments which modify channel substrate
    - Creation of gravel beds and other spawning habitats
    - Current deflectors or low dams
    - Construction of shallow water areas (e.g. bays, graded banks)
  - Structures to reduce fish mortalities at abstraction points and outfalls
    - Construction of fish screens
- Large-scale actions
  - Installation of fish migration facilities
    - Installation of fishways, ramps, lifts, locks or by-pass channels
  - Construction of shallow-water berms or shallow bays
    - Excavating of substrate with wash buffering constructions
  - Opening rivers to adjacent gravel pits or other water bodies
    - Breaking of levees and construction of connections
  - Braided rivers and construction of islands
    - River realignment, e.g. dyke removal and shallow bank substrate extraction
  - Multistage channels
    - River realignment, excavating of flood berms and construction of point bars
  - Remeandering
    - River realignment and setting back of levees
  - Pollution control
    - Diversion of water, pollution control and treatment by technology or reduction of agricultural fertilisers or amelioration and treatment through direct intervention, harvesting animals or provision of riparian buffer zones
  - Integrated floodplain restoration
    - For example, setting back levees, full width floodplains, reconnection of relic channels and floodplain water bodies, creation of new floodplain structures, provision of submersible dams, water regime regulation

Germany. Similarly, Collares-Pereira *et al.* (2002b) used the endangered cyprinid *Anaecypris hispanica* (Steindachner, Cyprinidae) as the target for rehabilitation and conservation actions, which should lead to protection and recovery of other endangered endemic species in the Guadiana River in Portugal. Thus, recovering or protecting populations in rivers may enhance the whole riverine fish community, whereas selective stocking will only 'sustain', at best, single stocks. However, socioeconomic factors may in many cases constrain the degree of river rehabilitation. Nevertheless, even under severe restriction, e.g. by flood control, there is the possibility to improve habitat and consequently inland fisheries by adopting an integrated habitat improvement approach (e.g. River Thames case-study, Banks 1990; Hughes and Willis 2000).

Standing waters

One of the greatest threats to many standing waters is human-induced nutrient (especially phosphorus and nitrogen) input (e.g. Sas 1989) – the process normally termed 'cultural eutrophication' (e.g. Mehner and Benndorf 1995). Eutrophication is largely attributable to high point-source loadings in the past, and high nutrient input from arable land at present (Jeppesen *et al.* 1999). It dramatically accelerates the natural ageing of lakes. The results of this anthropogenic nutrient loading are symptomatic changes such as increased biomass of phytoplankton, increased water turbidity, reduced oxygen content, reduced water quality, and occasionally decreased lake volumes (see Mehner and Benndorf 1995; Smith 1998 for reviews). In European standing waters, the fish community generally shifts from a dominance of salmonids and coregonids to percids and subsequently to cyprinids with increasing trophic state from oligotrophic to eutrophic (e.g. Hartmann 1977; Persson *et al.* 1991, but see Haertel *et al.* 2002; Olin *et al.* 2002). Altogether, the changes due to eutrophication are in almost all cases socially undesirable and often interfere with human water uses, e.g. drinking water, boating or bathing (Mehner and Benndorf 1995). However, from the inland fisheries point of view, eutrophication is sometimes considered beneficial (Barthelmes 1981) because of increasing productivity with increasing degree of eutrophication (Bninska and Leopold 1990). Only at very high nutrient loadings does the system's productivity drop. The breakeven point is uncertain and lies somewhere between the eutrophic and hypertrophic states (e.g. Bninska and Leopold 1990). Whether inland fishermen view eutrophication positively or not depends on their main target species. For example, commercial coregonid fisheries or recreational salmonid fisheries would suffer from declining stocks when the ecosystem moves from the oligotrophic to the eutrophic state. Inland fisheries targeting piscivorous predators complain about eutrophication long before cyprinid fishermen reach their maximum potential (see Bninska and Leopold 1990). However, it is noteworthy that in some countries the very valuable freshwater piscivorous fish, pikeperch, reaches its maximum abundance in polytrophic or hypertrophic states (e.g. Barthelmes 1981). Therefore, in Germany commercial fishermen are at present complaining about natural re-oligotrophication processes because stocks of commercially valuable pikeperch are declining (see also Lappalainen and Pönni 2000). These relationships reveal that the general goal of society to reverse cultural eutrophication (Mehner and Benndorf 1995) could sometimes impose conflicts with the interests of inland fisheries. Generally, the reversal of cultural eutrophication can be deleterious to inland fisheries because it can reduce fish production and biomass, or change the species composition (Maceina *et al.* 1996; Ney 1996). Thus, in contrast to running waters various stakeholders may have very different perceptions about the direction of lake and reservoir management, which may cause serious conflicts. Inland fishers, however, have to accept that the societal goal of reducing the negative impact of cultural eutrophication is of higher priority when compared with a relatively small stakeholder group of freshwater fishermen.

There are several technological, chemical and biological measures that may help in restoring eutrophicated lakes (e.g. Cooke *et al.* 1993). Water quality can be improved by: (i) reduction of external loading of nutrients; and (ii) controlling internal ecological processes without controlling external nutrient loading ('ecotechnology'), or a combination of both (Benndorf 1995). Among the latter, biomanipulation (Figs 7 and 11) or the trophic cascade, top-down food web management, can be grouped into so-called ecotechnologies (e.g. Hansson *et al.* 1998; Drenner and Hambright 1999). The basic goal of biomanipulation as a tool in water quality management is greater water transparency due to reduction in phytoplankton density (Shapiro *et al.* 1975). This may be achieved by promotion of planktonic crustacean biomass and an increase in their body size (e.g. Mehner *et al.* 2001). However, zooplankton is usually exposed to planktivorous fish predation (top-down mechanisms, e.g. Brooks and Dodson 1965). Therefore, a substantial decrease in planktivore biomass below a critical level, where the herbivore community is released from a too high top-down predation rate, is one of the major prerequisites to manipulate successfully a planktonic community in standing waters (e.g. Carpenter *et al.* 1985; Lammens 1999). To reduce the biomass of planktivorous fish, several strategies have been applied (e.g. Drenner and Hambright 1999). Removal of planktivorous fish with large nets was frequently conducted, particularly in shallow lakes (Meijer *et al.* 1999; Perrow *et al.* 1999). An alternative or supplementary method to fish removal is to stock high numbers of piscivorous fish (e.g. Benndorf *et al.* 1988; Berg *et al.* 1997). Wysujack and Mehner (2001) found that in a stratified lake (Feldberger Haussee, Germany) piscivorous fish consumption exceeded the removal of planktivores (roach) with the seine net in certain years, indicating piscivorous stock density may play a significant role in effective biomanipulation (but compare Drenner and Hambright 1999). Enhancement of piscivorous fish stocks may not only increase the top-down control of planktivorous fish but also be beneficial to inland fisheries (Ney 1996). This particularly applies to selected countries of Europe (e.g. Germany, the Netherlands, Switzerland) where most inland fisheries target top predators (see Section 3 for references and Pitcher and Pauly 1998). Thus, biomanipulation has the potential to couple water quality and inland fisheries management because the concept can be easily communicated to the broader (angler) public. Enhancement of piscivorous fish may then lead to increased

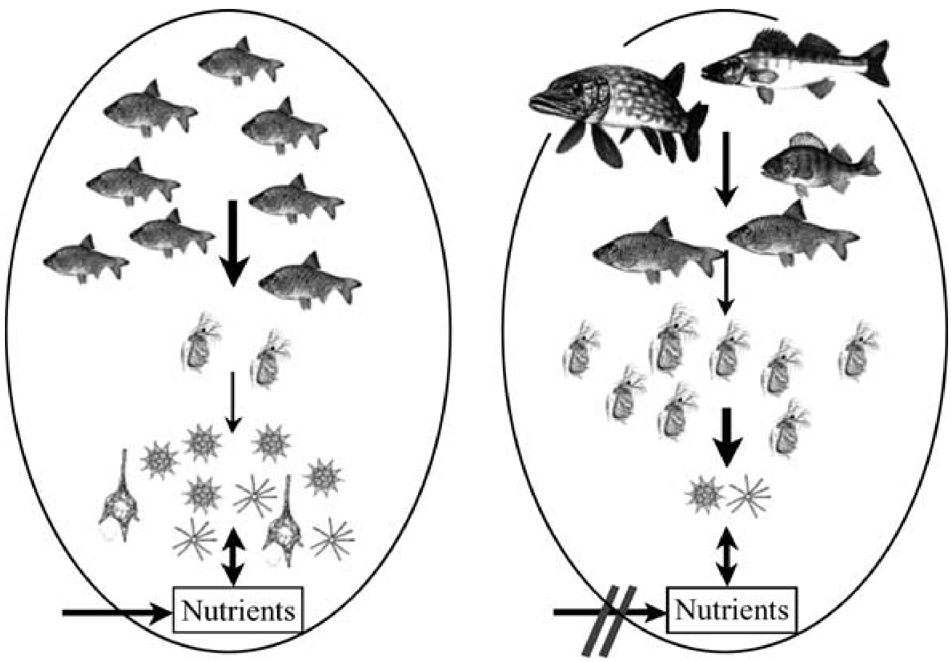

Figure 11 Scheme to demonstrate changes in the pelagic food chain in lakes during biomanipulation. Whereas planktivorous fish dominate and phytoplankton biomass is dependent on nutrient supply in eutrophic lakes without manipulation (left), piscivorous fish reduce planktivores and if external nutrient supply is blocked, grazing by daphnids reduces phytoplankton (right).

water clarity as well as more satisfied fishers and other outdoor recreationists (e.g. swimmers, boaters). Furthermore, in standing waters, where natural recruitment of piscivores is low, regular stocking may be necessary. This increases compliance within the traditional inland fisheries management system because many fisheries managers and inland fishermen believe that stocking is the most efficient management practice to enhance stocks (e.g. Wolos 1991; Klein 1996; Arlinghaus 2002). Nevertheless, there is a paucity of information to draw generally valid conclusions regarding the use of biomanipulation as a tool for eutrophication control (Benndorf 1995). The findings to date indicate that in shallow lakes the method will only have a long-term effect if (external) nutrient loadings are reduced to a level such that the total phosphorous (P) concentration is less than 0.05–0.1 mg P L^{-1} (Benndorf 1987; Jeppesen *et al.* 1999). Benndorf (1995) reported a 'biomanipulation efficiency threshold' of 0.5 and 2.0 g m^{-2} $year^{-1}$ total phosphorous loading. Furthermore, fishing and angling can have strong effects on fish populations of piscivores and food webs (e.g. Carpenter *et al.* 1994; Johnson and Carpenter 1994; Post *et al.* 2002) and preclude the success of lake rehabilitation through biomanipulation. Last but not least, if fishers and anglers do not agree with regulations, then regulations and biomanipulation are unlikely to work (Carpenter and Lathrop 1999).

One mechanism that has not received much attention to aid biomanipulation and fisheries management in standing waters is rehabilitation of the physical habitat. This is only practical when pressures from other users have eased or as a mechanisms to ameliorate a bottleneck in the fishery (piscivorous fish) recruitment processes. Rehabilitation of habitat include such actions as shoreline development, e.g. reinstatement of riparian vegetation, and creation of artificial and quasi-natural spawning grounds (Winfield *et al.* 2002; Zalewski and Frankiewicz 2002). Artificial reefs, made up of, for example, old tyres, artificial submerged habitats formed by spruce trees and replanting of submerged and emerged vegetation are also pertinent (e.g. Skov and Berg 1999; Sandström and Karas 2002). However, the scope for physical modification is limited and many of the rehabilitation practices are linked to the improvement of water clarity and quality discussed earlier.

To sum up, in Europe several successful restoration projects in shallow (e.g. Jeppesen *et al.* 1999), as well as large (e.g. Kairesalo *et al.* 1999; Suoraniemi *et al.*

2000) and stratified lakes (e.g. Mehner *et al.* 2001), indicate that a combination of external nutrient load reduction and biomanipulation enhancing pisicvorous fish stocks may, under certain conditions (Mehner *et al.*, unpublished data), lead to both increased water quality as well as enhanced compliance, satisfaction and benefit of the inland fishery system.

Synthesis

The last section synthesizes the issue of sustainable management of inland fisheries. Major challenges as well as perspectives and constraints for inland fisheries management are presented in a holistic manner. Because research produces knowledge and understanding, identifies issues needing attention, helps resolve conflicts and suggests solutions and new options (Williams 1998), essential needs for sustainable inland fisheries management will be presented with special emphasis on research. Key points are highlighted in italics.

Challenges – conservation and philosophical shifts

Inland fisheries management is today more a *multidimensional conservation* (e.g. Smith 1986; Olver *et al.* 1995; Mangel *et al.* 1996) than an allocation issue (e.g. Smith 1986; Loomis and Ditton 1993) that has to balance human requirements against protection of the environment and biodiversity (Cowx 2000). Modern conservation challenges for fisheries management encompass all aquatic resources within the *whole ecosystem* but also the *fishery per se*. One of the major challenges for future inland fisheries is to make sound management decisions to ensure viable commercial and recreational fisheries are compatible with aesthetic and nature conservation values in the 21st century (Radonski 1995; AFS 1999; Pitcher 1999). However, this requires harmonization of philosophical world views of rather biocentric (e.g. environmentalists) and anthropocentric (e.g. inland fishermen) orientated stakeholders (e.g. Rahel 1997), which resembles a sociocultural and political issue. The task for the future will be to apply the *stakeholder approach* (first management philosophy) to decision making with respect to freshwater systems because this philosophical shift for fish, wildlife and water resource management recognizes a larger set of beneficiaries of management (including the public and, in concept, future generations) (Decker *et al.* 1996; Nielsen *et al.* 1997). The key to improving implementation will include (compare Figs 5 and 10): (i) expanding the manager's view of who is substantially affected by fish and wildlife management (stakeholder); (ii) identifying and understanding stakeholder views; (iii) seeking *compromise* between competing and conflicting demands when appropriate (i.e. without risking the long-term integrity of fishery resources, but see Scheffer *et al.* 2000) and (iv) improving *communication* between managers and stakeholders (Decker *et al.* 1996). Ultimately, due to the expanded notion of values such as *responsibility* (for the fisheries resources), *fairness*, *justice*, and long-term *concern for the sustainability* of resources, the stakeholder approach forces inland fisheries managers to consider ethical questions in decision making (Decker *et al.* 1996; Williams 1997).

The impact of philosophical shifts in inland fisheries resourc management goes further. It encompasses a shift from the sectoral to the *system view* of inland fisheries management – away from single species to *multispecies* and *ecosystem-based management* (second management philosophy), being aware that single species are nested elements of ecosystems linked to the environment including man. However, *participation* and *involvement* of *all* stakeholders in *local decision making* remains the key element of this global philosophical shift because sustainability integrates not only ecological but also social and economic dimensions. EBM determines that sustainable management of inland waters is *integrated or holistic* management of watersheds or specific aquatic ecosystems (see Scheffer *et al.* 2000 for a theoretical analysis). However, most of the factors causing problems for fish communities and fisheries lie outside the control of the inland fisheries management system (Cowx 2000). Furthermore, it is well known that in many cases, those operating in the broader water resource planning sector seldom, if at all, actively solicit the input of fisheries experts and managers (Cowx 2000). Consequently, on a larger scale (e.g. watershed management), major challenges for inland fisheries managers and specialists are: (i) to bring in and defend the *interests* of the fisheries stakeholders (e.g. fishers and anglers; compare successful case-studies Chandler 1990; Pajak 1992) by, for example, interacting and making alliances with other interested parties; (ii) to seek to *limit damage* to aquatic ecosystems and (iii) to promote *rehabilitation* activities (Cowx 2000). On a smaller scale (e.g. local management of small gravel pit of an angling club), a major challenge of the local fisheries manager is to adopt widely the *precautionary approaches* and *principles* (e.g. when stocking) to make socially and ecologically 'safe' and reversible decisions. Giving the trend of (urban) recreational fisheries towards concentrating on intensively stocked artificial fisheries, this might release pressures on natural fisheries for a less heavily exploited approach. Precaution, however, should accompany every management decision concerning natural resources and the environment. To become sustainable, inland fisheries management has to reduce the degree of arbitrariness in decision making. Integration of *evaluation* procedures and the implementation of a *thorough fisheries management process* (Fig. 10; Van Densen 1990; Taylor *et al*. 1995; Brown 1996; Krueger and Decker 1999) is an essential condition for sustainable management and the application of *adaptive management systems* (e.g. Walters 1986; Smith and Pollard 1996).

Basic needs with emphasis on research

Basic needs to approach sustainability in inland fisheries are outlined below. All of the issues described satisfy one of the major axes of the stakeholder satisfaction triangle – substance (technical and factual content of the situation revealed, e.g. by science or experience), process (steps to follow in a management decision) and relationships (development of positive networks among individuals with direct or indirect interest in or influence over a management decision) – that together determine satisfactory decision making in future inland fisheries management (see Nielsen *et al*. 1997; Meffe 2002 for details).

Communication and information

In fresh waters (and in all management issues all over the world), there is an urgent need to improve communication and information links between managers, scientists and various stakeholders (e.g. anglers, swimmers, fundamental environmentalists, Loftus 1987; Decker and Krueger 1999; Hickley and Aprahamian 2000; Ludwig 2001; Meffe 2002). This should be the first step in management of the resource, especially in larger scale systems (e.g. river fisheries on the catchment scale) (Ostrom *et al*. 1999; Cowx 2000).

Education and actor empowerment

Peoples' abilities to know and act must be developed if we are to sustain progress in inland fisheries management (e.g. Taylor *et al*. 1995; Schmied and Ditton 1998; Williams 1998; Ludwig 2001). There is a need for lifelong education and training to build human capacity and actor empowerment (Williams 1998). Here also lies a crucial role for research to develop regular educational programmes and to train the public, anglers (Von Lukowicz 1998) and (voluntary) fisheries managers (Walder and Van der Spiegel 1990; Brown 1996; Williams 1998; Hickley and Aprahamian 2000; Rassam and Eisler 2001; Cambray and Pister 2002; Meffe 2002). Scientists can be most effective if they make their results accessible to laypersons (Ludwig 2001). Such education outreach should result in more realistic expectations of fishery stakeholders about expected outcomes in degraded and often heavily exploited freshwater ecosystems. Educational needs for future fisheries managers (students) include integrated thinking about biological, physical, chemical and sociocultural processes; problem solving (conflict management, decision analysis) and communication skills (see above). Furthermore, education of fishermen should: (i) yield awareness of ecological carrying capacity and responsibility for ecosystem health; (ii) result in interest in the future sustainability of the fishery resources and (iii) keep expectations reasonable and more in accord with available and fluctuating fish resources (Hudgins and Davies 1984; Smith 1986; Gale 1992). Because it seems virtually impossible to increase fish populations to meet an ever-expanding demand, it is reasonable that reduced expectations will produce greater fishing satisfaction, particularly in recreational fisheries (Graefe and Fedler 1986; Spencer and Spangler 1992).

Institutional restructuring

New institutional (e.g. Holland 1996; Ostrom *et al*. 1999) and management structures incorporating specialized personnel are needed. This should lead to drastic restructuring in public authorities (Cochrane and Payne 1998) to provide institutional linkages between, for example, fisheries, environment, water, nature conservation or animal welfare organizations and public agencies (Meffe 2002; but see Ascher 2001 for perverse dynamics created in high-level institutions). Moreover, groups of inland fishers and anglers who can communicate and identify with one another are more likely than groups of strangers to draw on trust (direct and indirect) reciprocity (e.g. Nowak and Sigmund 1998; see also Riolo *et al*. 2001 for cooperation without reciprocity), and reputation (Milinski *et al*. 2002) to develop norms and voluntary rules that limit unsustainable exploitation of common pool resources such as fish stocks. Punishment of noncooperating anglers and fishers can cause a rise in the level of the average contribution to the conservation of public good (e.g. Fehr and Gächter 2000), and is one means to enhance cooperation between fishery stakeholders to sustain the resource based on voluntary rules. These informal institutions are needed to solve common pool resources problems and unsustainable inland fisheries management (Ostrom *et al.* 1999; Ludwig 2001).

Marketing outreach

Many believe (e.g. in Germany) that commercial fisheries might be able to enlarge the quantity of fish harvested and sold because of the modern trend in agriculture caused by the 'BSE crisis' in the beef trade to bring products to more regional markets, which in inland fisheries is traditionally the way goods are marketed. However, in many commercial fisheries of developed countries, a marketing outreach is needed to attract consumers and anglers to their waters or even change consumer habits favouring selected fish species (e.g. cyprinids) or fish products (compare Sutton 1998). In the case of coexploitation of commercial fishing rights by commercial and recreational fisheries, newly recruited anglers might be a valuable source of income for commercial fishermen. In the case of difficulties in selling fish, commercial fishermen would be wise to act as service providers of the angling experience to a greater extent. Furthermore, a general marketing outreach (e.g. Salwasser *et al.* 1989; Janisch 2001) should yield support from the general public for fish and fisheries' interests (Alcorn 1998), and may recruit new anglers and fishers in the case of demographic change (Wilde *et al.* 1996; Fedler and Ditton 2000). The latter is a particularly important issue because the drop in participation in angling experienced in many European countries (Cowx 2002c) needs to be addressed. Anglers are considered in many countries as the guardians of the environment and the eyes and ears of the protection agencies. Without their presence on the rivers and lakes, the protection they afford will be lost, probably to the detriment of the environment.

Management plans

When managing for sustainability, inland fisheries need the domination of local bottom-up management structures, and also some kind of top-down advice, education and control to better manage the 'unsustainable knowledge' of local (voluntary) fisheries managers, e.g. of a small angling club, and educate local fisheries managers in the long term. Management plans formulated by local fishermen and fisheries managers and controlled and enforced by professional fisheries experts and scientists in management agencies provide a means to link institutionally top-down (legal framework, public authorities) and bottom-up (local stakeholders, mainly fishers and anglers) traditional inland fisheries management (e.g. Souchon and Trocherie 1990; Hart and Pitcher 1998; Knösche 1998). However, to set up a thorough management plan is a complex task; and it is doubtful whether, for example, those participating in rod and line fisheries have the ability to assess the stock accurately (Cowx 1991, 1996, 2002b; Walters 1998). Thus, a stronger, hand-in-hand, cooperation of commercial and recreational fisheries is needed to bridge the gap of 'economically endangered' commercial inland fisheries and 'ecologically endangered' recreational fisheries which may lead to self-regulated and effective (sustainable) fisheries management systems (Hart and Pitcher 1998; Scott 1998; Ostrom *et al.* 1999). However, it is necessary to develop simple guidelines about the essential procedures required to set up a management plan.

Decision analysis

Public agencies should be prepared to use science-based, decision-support techniques (e.g. from the field of operations research, e.g. Lane 1992), such as: (i) several variants of multicriteria decision analysis (e.g. Healey 1984; Saaty 1990; Merritt and Criddle 1993; Merrit and Quinn 2000); (ii) statistical decision theory (e.g. Bayesian decision analysis, Heikinheimo and Raitaniemi 1998; Peterman *et al.* 1998; Robb and Peterman 1998); (iii) multivariate statistical methods, e.g. multidimensional scaling (Pitcher *et al.* 1998; Pitcher and Preikshot 2001); (iv) broad benefit/cost analysis (Talhelm and Libby 1987; Bilsby *et al.* 1998; Scheffer *et al.* 2000); (v) graph-theoretical techniques (Brüggemann *et al.* 2001) or (vi) several variants of computer simulation models (e.g. Johnson *et al.* 1992; Cole and Ward 1994; Johnson 1995; Radomski and Goeman 1996; Pan *et al.* 2001; Cox and Walters 2002b), to improve decision making and judgement in inland fisheries management. These tools of decision analysis aim mainly to incorporate the complex nature of sustainability and fisheries management to select the best case scenario; that is the management option that maximizes societal welfare without compromising the aquatic ecosystems.

Human dimensions research and social sciences (excluding socioeconomic evaluation)

Fisheries management is increasingly seen to be as much about managing people as about fish stocks (e.g. Pringle 1985; Barber and Taylor 1990; Clay and McGoodwin 1995; Jentoft 1998, 1999; Pauly *et al.* 1998a). In the past, however, social sciences rarely played a role in fisheries management (Clay and McGoodwin 1995; Jentoft 1998, 1999) and most analysis of the human aspects of fisheries were nonquantitative, with little predictive or diagnostic power (Pitcher *et al.* 1998). Consequently, there is an urgent need for sophisticated human-dimension research and for social sciences in inland fisheries management (e.g. Brown 1987; Talhelm and Libby 1987; Peyton and Gigliotti 1989; Loomis and Ditton 1993; Decker and Enck 1996; Ditton 1996; Wilde *et al.* 1996; Enck and Decker 1997; Aas and Ditton 1998; Harris 1998; Jentoft 1998). Incorporating this knowledge into models should allow a better understanding of human behaviour (Anderson 1993; Gillis *et al.* 1995; Radomski and Goeman 1996; Provencher and Bishop 1997; Smith 1999). With respect to recreational fisheries management, European fisheries managers and researchers have to realize that more quantification of angler values, preferences and behaviour is needed, along with greater scientific experimentation of regulations, such as creel and length-based limits (Wilde 1997), to optimize angler satisfaction or benefit (OSY, Radomski *et al.* 2001).

Socioeconomic evaluation

For inland fisheries managers, socioeconomic aspects of mitigation and rehabilitation (including finance) are currently a more problematic feature than technical aspects, and present one of the greatest challenges to development and maintenance of inland fisheries and aquatic ecosystems (Lucas and Marmulla 2000). This calls for thorough socioeconomic evaluation of inland (especially recreational) fisheries (e.g. Brown and Knuth 1991; Hickley and Aprahamian 2000; Hughes and Morley 2000) to ensure that they are well represented in all development activities concerning freshwater ecosystems (Cowx 1999a, 2002a). The numerous benefits inland fisheries provide to society have to be investigated to make the many intangible benefits of inland fisheries quantifiable and objective.

Traditional fisheries science

Without basic biological information on, inter alia, fish stocks, exploitation level, harvest and habitat, there can be no credible management planning (Van Densen 1990; Cowx 1991, 1996, 2000, 2002b; Quinn and Szarzi 1993; Johnson and Martinez 1995; Radomski and Goeman 1996; Haggan 1998). This information is lacking with respect to recreational fisheries in many European countries (compare also Smith and Pollard 1996; Post *et al.* 2002), but is easily accessible by routine monitoring of the fishery, e.g. by creel surveys (Guthrie *et al.* 1991; Pollock *et al.* 1994) or a combination of creel and intercept surveys (Ditton and Hunt 2001). Moreover, despite widespread adoption of stocking and introduction practices throughout the world, there is surprisingly little information about success, economic efficiency, ecological effects of stockings and the way stocked systems function (Cowx 1996, 1998b; Welcomme 2001). Therefore, there is a need for post stocking monitoring programmes and feedback to the public domain (Fig. 8), so more rational evaluations of the outcomes of stock enhancement procedures can be made.

Aquatic ecology

Future ecological research must strive to identify and understand the mechanisms (spatial and temporal), dynamics and processes driving large-scale ecosystems (Neill 1998; Parsons *et al.* 1998; Scheffer *et al.* 2000, 2001) and fish community changes (Rose 2000; Jackson *et al.* 2001). It is desirable to develop ecosystem models that can predict whole-community changes (Pitcher and Pauly 1998; Jackson *et al.* 2001), and fisheries managers should be trained to use them (Giske 1998). Although there are many aquatic ecosystem trophic models available (e.g. ECOPATH, ECOSIM or ECOSPACE, see Whipple *et al.* 2000 for a review), these models have to be tuned to allow the setting of unambiguous, operational objectives in ecosystem and fisheries management (e.g. rebuilding ecosystems, healthy ecosystems). However, an urgent need remains to integrate ecology with the dominant top-down component of most freshwater ecosystems – the human dimension and socioeconomics – to understand (and manage) ecological patterns and processes on a sustainable basis (Costanza 1996; Liu 2001).

Perspective

Many societal, political and environmental trends promote the development of sustainable fisheries. For example, EBM and sustainability are concepts widely accepted and adopted by societies of the industrialized world. Moreover, since the early

1980s, in Europe the state of many freshwater ecosystems has improved remarkably and basin-wide river and lake management is nowadays a common approach. In addition, the European Water Framework Directive (WFD) provides an excellent opportunity to improve the quality of freshwater ecosystems and therefore ensure the (ecological) sustainability of inland fisheries (Pollard and Huxham 1999). In future, the biology (and not just the chemistry) of each water body will be the key criterion for protection and rehabilitation activities. The legal instruments of each member country at the national and federal state levels must be promulgated so that the demand for 'good ecological quality' can be attained. Exceptions may be only conceivable in the case of 'heavily modified water bodies' which should yield a 'good ecological potential'. The WFD demands that all stakeholders and the public should participate in development, evaluation and updating of management plans (Article 14 of the WFD). Besides, the WFD offers the opportunity for commercial fishermen to achieve a perpetual source of income by contributing to the regular monitoring of the fish community (compare Hart and Pitcher 1998), which should be conducted regularly by member states (Article 11). This might even lead to a new role for commercial fishermen.

In addition to favourable societal, political and environmental developments, many dimensions of the inland fisheries system are also propitious for the sustainable development of inland fisheries. For example, bottom-up driven traditional management, small-scale structures, well-developed (group) property rights, and the predominance of selfish benefit-maximizing recreational fisheries constitute excellent conditions for a better management (compare Table 3 and Ostrom *et al*. 1999). However, environmental degradation, low social priority, serious conflicts between user groups, relatively low funding for fisheries research and management, inadequate enforcement and control, and low education level, inter alia, constrain the future existence and development of inland fisheries (compare Table 3). Nevertheless, the sustainability debate *per se* provides a chance for the adoption of many of the issues mentioned throughout this paper and challenges all stakeholders to move towards sustainable inland fisheries.

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