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INTRODUCTION

Fishing always has been, and for the foreseeable future will remain, a major source of food and income for society (Cowx 2002c). However, its importance relative to other food production systems has evolved, especially over the last half century, as a result of the way fisheries are exploited (FAO 1997). This is especially true of fishery activities in inland waters with different scenarios being enacted in the densely populated and highly industrialised countries of the northern temperate world and tropical developing countries (Arlinghaus, Mehner and Cowx 2002). These differences are largely the result of contrasting social and economic objectives for inland fisheries and the different ways they are managed (Table 1, after Welcomme 2001). Fisheries management in industrialised countries focuses almost exclusively on recreation and conservation, whereas the objective in developing countries remains largely on food security, although a shifting emphasis towards recreational fisheries (Cowx 2002c) and conservation (Collares-Pereira, Cowx and Coelho 2002) is occurring as a result of globalisation and the influence of international protocols such as the Convention for Biological Diversity.

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

Industrial (temperate)	Emerging economies (tropical)
Objectives	
Conservation/Preservation Recreation	Provision of food Income
Mechanism	
Recreational fisheries Habitat rehabilitation Environmentally sound stocking Intensive aquaculture	(Commercial) Food fisheries Habitat modification Enhancement, e.g. through intense stocking Extensive, integrated, rural aquaculture

Economic	
Capital intensive	Labour intensive

This diversification of objectives arises because increasing exploitation of inland fishery resources, both in terms of effort and fishing efficiency, tends to reduce opportunities for production of fish as food and shifts resource use towards recreational needs (Smith 1986; Radonski 1995). Consequently, in most temperate countries recreational, leisure or "sport" fisheries are the dominant components of inland fisheries systems that evolved from a simple food production focus (FAO 1999; Welcomme 2001; Cowx 2002c). In developing countries food security and employment remain the primary focus (FAO 1997) despite major changes in aquatic resource use in these countries.

In addition, multi-purpose use patterns in industrialised countries have created a very distinct climate for the development of inland fisheries (FAO 1997). Activities such as agriculture, damming, flood control, deforestation, navigation, wetland reclamation, urbanisation, hydropower generation, water abstraction and transfer and waste disposal (Cowx 2002a) 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 industrialised countries are considered impacted (Dynesius and Nilsson 1994; Vitousek *et al.* 1997). Similar diversification of aquatic resource use is prevalent in developing countries (Nguyen Khoa, Lorenzen and Garaway 2003; Nguyen Khoa *et al.* 2003), but the impact is less dramatic and fishing for food has remained a sustainable activity, although fisheries are also under threat from development and shifts in fishery management activities to support production from culture based fisheries (Araujo-Lima *et al.* 2003; Pusey 2003) and aquaculture are occurring (van Brakel, Muir and Ross, 2003). In this context, capture and culture fisheries must be seen as complimentary and not alternatives (van Brakel *et al.* 2003), as this could potentially lead to reduced production from a particular water body. Conventional aquaculture is also not necessarily an option for the rural poor and diversion of resources from capture fisheries could contribute to food insecurity. Aquaculture in these circumstances should be focussed on enhancement of natural production of resources accessible to the rural poor.

Basically, inland fisheries can be viewed as evolving organisms (Figure 1), with the major stages in the life cycle of an inland fishery comprising an initial phase on food production, then a growing interest in recreation, with aesthetic and nature conservation interests emerging last (Smith 1986). Although this process is a continuum, industrialised countries can be envisaged at one end of the spectrum and developing countries towards the other, depending of the scale of industrialisation that has taken place. This is, however, a simplification because the need for food security has triggered activities such as aquaculture and fish stock enhancement strategies (culture based fisheries), which replace or support fish production, especially in developing countries (see Petr 1998 for review). Thus, in most areas of the world the principal impacts on inland fisheries do not originate from the fishery itself but outside the fishery (e.g. FAO 1997; Garcia, Cochrane *et al.* 1999; Welcomme 2001). The need for concerted effort to prevent and reduce degradation, as well as conserving freshwater fish and fisheries as renewable common pool resources or entities in their own right, are the greatest challenges facing sustainable development of inland waters (FAO, 1999).

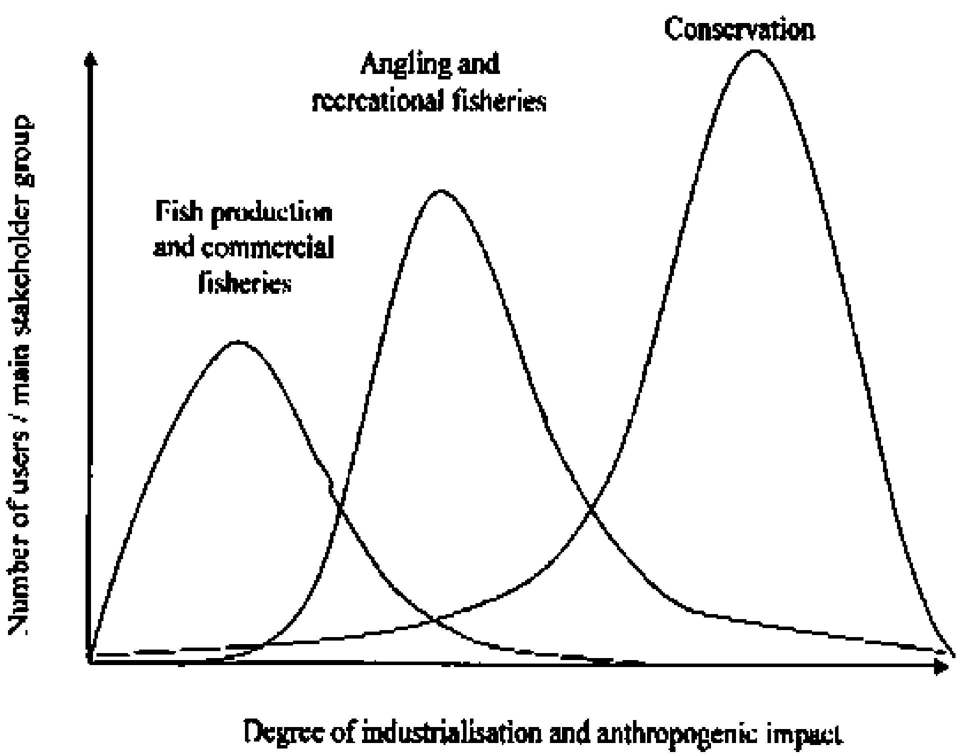


Figure 1. Generalised evolution of inland fisheries along an industrialisation gradient (modified from Smith 1986).

One of the underlying aspects relating to these changes that has received little attention is the value of the fisheries and aquatic resources, including the importance of ecosystem services and biodiversity. Fisheries are poorly or undervalued in multiple aquatic resource-user scenarios and this has undoubtedly contributed to the changes described (Cowx 2002c). However, it must be recognised that in some rural communities, fisheries are considered of little importance and thus of minimal value and it is important to understand why fisheries are valued differently between these locales (Bene and Neiland 2003). This paper examines the importance of accurately valuing the fisheries of large rivers (and in all ecosystems) and how such information can be used to maintain, improve and develop inland fisheries and ecosystem services, from both the exploitation and conservation perspectives, for future generations.

TRENDS IN INLAND FISHERIES

Although the net contribution of inland fisheries to total world fish production is small in comparison to marine capture fisheries and aquaculture (Figure 2a), it has sustained a growing trend of about 2 percent per annum, worldwide (FAO 2002). However, this growth belies the true picture elucidated from a regional review (Figure 2b). Net declines in catches are prevalent in Europe, much of which occurred post decentralisation of the eastern European economies and North America. The main increases have been in Asia and Africa, the latter being mainly due to increased yield from lakes, especially by Nile perch, *Lates niloticus* (L.), from Lake Victoria. Production figures in Asia have increased for a number of reasons, notably the proliferation of culture-based fisheries in China and Bangladesh, but also because more reliable catch statistics data from, for example, the Mekong countries, show the true extent of exploitation. Notwithstanding these trends the overall picture for natural river fisheries is unfavourable. Throughout the world, there is no doubt a river fishery makes valuable contributions to leisure activities and food security, but their performance is generally on the wane or shifting in character.

Throughout the literature and electronic statistical databases, commercial/artisanal/subsistence catches from the major river fisheries generally indicate declining trends, which have direct implications for rural livelihoods (Bene and Neiland 2003; Hand 2003). By contrast, improvements in recreational/sport catches are evident in Western Europe and North America as a result of rehabilitation and restocking activities, e.g. River Rhine (Brenner, Buijse, Lauff *et al.* 2003). However, in both scenarios the fisheries and fish community structures are changing. Increasing fishing pressure usually results in a decline in yield but also fishing down of the food web whereby a marked shift in catch composition toward individuals and species that mature at a small size and/or age is observed (e.g. Oueme Delta fishery, Welcomme 2001). These early maturing, small-sized fishes tend to be economically less valuable and less desirable than the large predatory species that are removed from the fishery first. Notwithstanding this argument, these small-sized fish are nutritionally important and often contribute greatly to food security in rural areas. Similarly, enhancement of recreational fisheries through stocking and introductions has altered fish community structures markedly toward species considered desirable by anglers, often to the detriment of the indigenous species (Cowx 2002c). These changes in the fisheries structure and function not only have marked impact on the economic value of the fisheries, but also have considerable environmental cost (Arlinghaus *et al.* 2002).

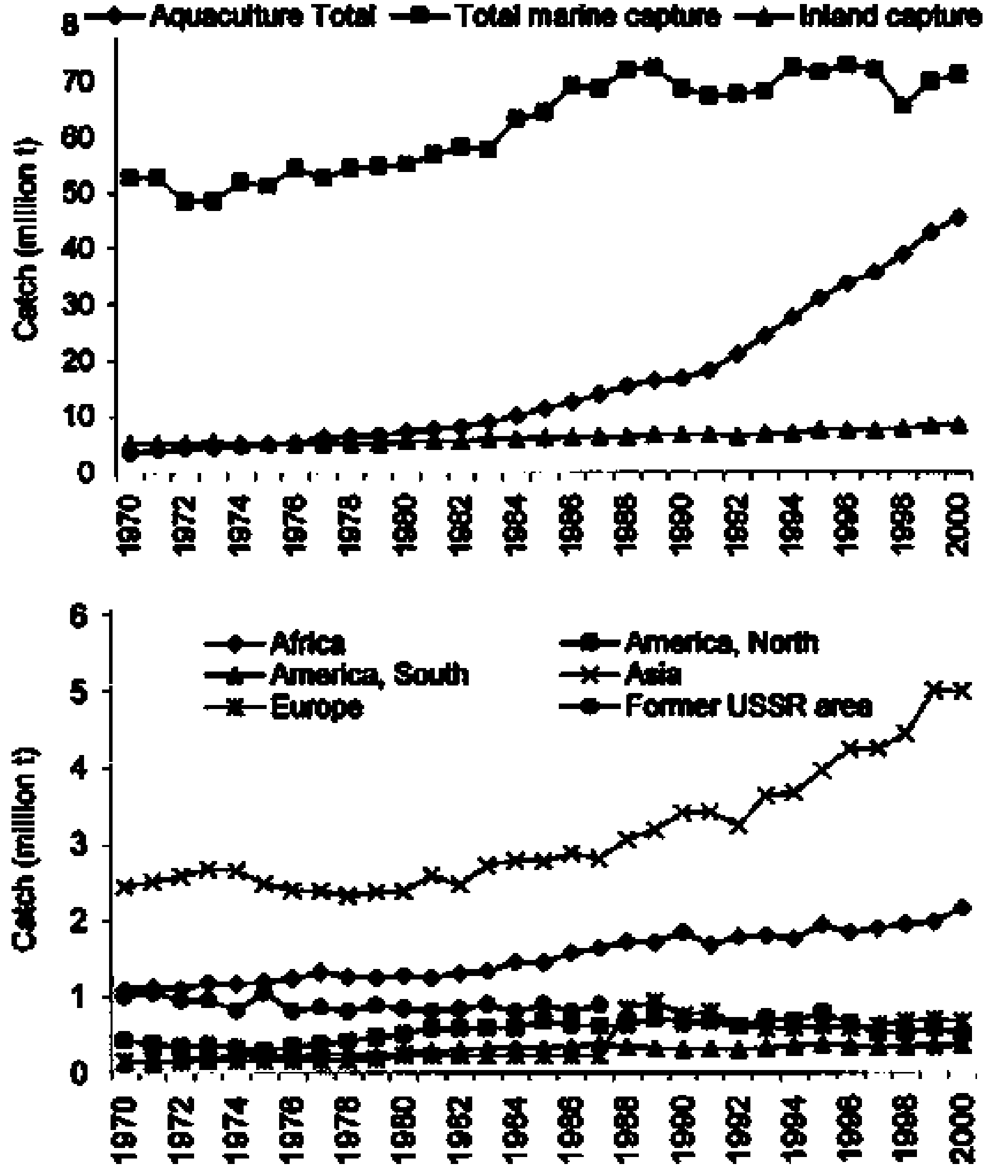


Figure 2. a) Trends in production from aquaculture and marine and inland fisheries between 1970 and 2000 and b) catches from inland waters by continental region (European data for 1988 onwards include former USSR data) (Source: FAO 2002). Note, that underreporting of catch by countries and incomplete data on recreational fisheries suggest that catches in inland waters may be at least twice as high as shown in the figure (FAO 1999).

WHY VALUE ENVIRONMENTAL GOODS?

Despite recent developments in inland fisheries, they undoubtedly have high socio-economic and socio-cultural importance and provide "a myriad of benefits to society" (Weithman 1999; Welcomme and Naeve 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). There are a number of reasons for this, including the fact that it is difficult to assign value to such factors as the value to artisanal and subsistence fisheries of reducing risk and vulnerability to poverty or the high social and cultural value, often in terms of community solidarity, festivals and spiritual links, especially in agrarian countries. This problem, however, should not be used as an argument for not valuing the benefits of inland fisheries. Three main arguments exist for pricing environmental goods like river quality and protecting aquatic ecosystem function and biodiversity, including river fisheries (after Navrud 2001).

First, some socially optimal quantity/quality of an environmental good exists where the marginal cost of supplying the good is equal to or less than its marginal benefit, expressed as the public's demand for the good. This argument stems from the increasing awareness among policy makers that a non-zero goal of environmental degradation has to be accepted and that trade-offs can be viewed in economic terms as costs and benefits. Although the costs of supplying environmental quality (usually the costs of protection and rehabilitation) is relatively easily to determine, the demand for environmental quality in terms of corresponding benefits is more difficult to value. For example, improving the quality of effluent discharge into rivers will reduce pollution and increase the diversity and potential for exploitation of fish stocks. The same result can be achieved by stocking. While the social costs of pollution control programmes are relatively well known, the social benefits in terms of improved fish stocks are not. To calculate the optimal level of pollution control or optimal stocking regime there is a need to know the social benefits. To conduct a complete cost-benefit analysis of pollution control, the social benefits from all environmental improvements have to be calculated. Thus, not only increased fish stocks, but also reduced costs of treating water supplies for agricultural and human uses, reduced health impacts, increased uses for recreational purposes and other damage to estuarine or marine ecosystems etc., have to be valued.

Environmental goods have significant public good characteristics since individuals generally cannot be excluded from enjoying environmental improvements nor can they avoid environmental degradation. Thus, these goods are generally not bought and sold in markets and have no market prices or have market prices that do not reflect the full, marginal social costs of providing them. However, there is a need to know the marginal values or prices of environmental goods to be able to compare marginal costs and benefits and set an economic efficient level for the provision of these goods and corresponding environmental policy goals. Environmental prices are also needed to decide upon which regulations and projects are socially most desirable. Monetary values enable alternatives to be ranked and, by reference to other uses of the resource being valued, enable the opportunity cost (the value of opportunities foregone in order to derive a benefit) of each option to be compared.

Second, if environmental goods are not valued explicitly, policy decisions will value them implicitly, which often produces an arbitrary and inconsistent set of prices, because decision makers are often unaware they make these valuations. To illustrate this argument, consider a hydropower development project. This could pose environmental impacts on recreational and/or commercial fisheries, outdoor recreational activities, agriculture, forestry, water quality and supply, cultural and historical objects, landscape aesthetics and the ecosystem in general, which are usually only defined in a qualitative manner. Rarely is any attempt made to value these environmental impacts. Cost-benefit analyses (CBA), taking into account all social benefits and costs except the environmental costs, tend to be based on the net present value. If the net present value of a dam designed to be operational for 50 years is $10 million, with a discount rate of 5 percent pa, the annual net benefits are about $700 000. If the construction of the dam has consequences only for a local community of 20 000 persons, policy makers have implicitly valued the damage such that each person is willing to pay less than $35 each year to avoid the environmental damage. This value is much less if the river is of national or international significance and many people are affected. However, people have not been asked about their preferences and might be willing to pay more than this amount to avoid the negative environmental effects and preserve rivers (Navrud 1994). Consequently, the total social costs of the hydropower project will most likely exceed the benefits. Thus, from an economic point of view the dam should not be constructed. Care must be paid when adopting this type of valuation as there is a great deal of uncertainty when transferring benefits or costs from a study site to the policy site (i.e. the site for which values are needed). However, this uncertainty is judged to be acceptable in cost-benefit analysis, as other benefit and cost factors could easily be equally or more uncertain (Navrud 1994).

Third, there is the need to promote inland fisheries and aquatic biodiversity in environmental impact assessment and conflict resolution situations. This need largely falls out of the previous argument about decision makers making implicit valuations. When decisions are made on major development schemes the fish and fisheries must present a strong economic argument, otherwise they are overridden and suffer in the face of economically strong sectors such as hydropower production, water supply and agricultural development (e.g. Halls, Shankar and Barr 2003; Kaunda and Chapotk 2003). In many respects this argument is the most important because of the recognition that inland fisheries is just one element of a multi-user environment and the sector often promotes a weak argument for sustainability because it is undervalued in real terms. This issue is discussed in more detail later.

Despite the arguments presented above, there are some objections to assigning values to natural systems. The first objects to the role of market prices, implying the consumers are the best judges of the value of a system and that community considerations are irrelevant. This may be countered by noting the purpose of valuation is to provide more information to the political process of resource management, rather than leaving the process to be influenced wholly by political considerations. The second problem is the assumption that consumers understand the value of the ecological services provide by biological resources.

The objection revolves around the complexity of ecological process and, therefore, the need to treat the system as a whole. One implication is that if there is a case for the protection of a system, great care has to be taken in an even moderate level of use because of the frequent lack of knowledge about what species or parts of the system are necessary for ecosystem maintenance and what are redundant. For example, the ecological processes associated with aquatic vegetation are necessary to provide the appropriate conditions for favourable fish habitats. It is frequently difficult to assess, however, what extent of vegetation cover is necessary to provide for this favourable status. The third problem is that the techniques used to assess economic value tend to ignore many equity and moral considerations. The economic argument is that the techniques are quite distinct from the political recognition that inequities exist and of the need for the necessary policy instruments to reduce or remove them.

BENEFITS VALUES AND IMPACTS

Generally, three domains can be distinguished where benefits associated with river fisheries are accrued, *viz*.: economic, social and ecological benefits (Table 2). Furthermore, when reviewing impacts additional components need to be taken into account: (a) negative impact of fisheries on aquatic ecosystems; and (b) impacts, threats and constraints on river fisheries.

Table 2: Socio-economic benefits of inland fisheries and impacts on inland fisheries (modified from Weithman 1999)

Values	Impacts
Economic benefits	
Direct use: Consumptive, non-consumptive, indirect option Non-use: Existence, bequest	Direct, indirect, induced
Social benefits	
Cultural, societal, psychological, physiological	Quality of life, social well-being
Ecological benefits	
- Species diversity - ecosystem goods and services - maintenance of habitat	Mitigation, rehabilitation, management, negative "benefits" (impacts) Other impacts environmental degradation, low societal priority, user conflicts, cost-effectiveness, constraints

ECONOMIC BENEFITS

Total economic value (TEV) of river fisheries can be divided into two main components: a) direct use value; and b) non-use/preservation value. TEV is the sum of all use and non-use values, no matter how derived.

The direct use value of a fish stock can be divided into consumptive, non-consumptive and indirect values (Randall 1987; Bishop, Boyle and Welsh 1987; Table 2). Consumptive use includes the net income from commercial fisheries (i.e. income from fish sales minus the cost of input factors), harvest by an angler, or the economic value of recreational fisheries. This is the main criteria used to value fisheries in both industrial and developing countries, see for example Almeida, Lorenzen and McGrath (2003). Non-consumptive use (value that individuals derive that is not conditional on consumption of, or physical change in, natural resources) includes research or sightseeing, for example salmon jumping up a waterfall, fresh air and other public goods that do not deplete the fishery resources. Indirect use (also referred to as ecological function values) comprise all the ecological functions within a system, or may include activities away from the site (i.e. not fish used directly for food or sport), including trade, reading about or special activities at the fishery location (Riechers and Fedler 1996). A derivation of direct use and indirect use values are option values (value to an individual of maintaining the option to use a resource some time in the future). These may be seen as extra insurance against the risk of losing goods and services that are important in the life of the community. As such, they are also part of preservation values (see below).

Non-use values are the values which can be attributed to systems as a result of certain people deriving satisfaction from simply knowing that certain systems exist, although they do not obtain any direct or indirect goods and services from it. Non-use value is partitioned into 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. Riechers and Fedler 1996; Weithman 1999; Peirson, Tingley, Spurgeon *et al*. 2001). Existence values are, perhaps, more widespread among industrialised countries (in the UK, for example, the Royal Society for the Protection of Birds is the top income generating charity based on donations and directs its income to habitat protection), but not entirely. There also exist preservation values that are similar to option values. These are values attached by those who benefit directly or indirectly from preserving and natural system. This is a value to communities that fishing accords (e.g. McGrath, Castro, Futemma *et al.* 1993; McGrath, Silva and Crossa 1998). In this situation it is possible to create a market for environmental services, such as carbon sinking, reduction of erosion and control of fire.

Additionally, if a project could lead to irreversible impacts such as the extinction of a fish species, a quasi-option value may be assessed and used as a correction factor to the total economic value. This is equivalent to the precautionary principle and relates to the value of increased information about the value of fish species gained from not implementing the project with irreversible impacts. Another related concept is the Safe Minimum Standard (Bishop *et al*. 1978), which says preserve unless the costs are intolerable. The challenge is to define how high the costs can be before they become intolerable.

There are two main approaches to valuing non-market goods: i) methods based on individual preferences; and ii) methods based on preferences of experts and decision makers. The latter includes methods like multi-criteria decision analysis, Delphi techniques and implicit valuation elicited from political decisions (see above). These methods can be viewed as complementary decision tools to cost-benefit analysis (CBA) and will not be discussed further. Valuation techniques based on individual preferences can be broken down into two approaches: revealed preference and stated preference methods (Table 3).

Table 3: Classification of environmental valuation techniques based on individual preferences (modified from Navrud 2001)

	Indirect	Direct
Revealed Preferences	Household Production	Simulated markets

	Function (HPF) Approach - Travel Cost (TC) method - Averting Costs (AC) Hedonic Price (HP) analysis	Market prices Replacement Costs (RC)
Stated Preferences (SP)	Contingent Ranking (CR) Choice Experiments (CE) - Conjoint Analysis	Contingent Valuation (CV)

Revealed preference methods use data from observed behaviour of respondents in markets related to the non-use value. The Travel Cost method assumes that the costs incurred travelling to the recreational site (including direct travel costs, accommodation and expenditure on food etc.) are a complementary good to recreational activities. The basic premise of the method is that the number of trips to a recreation site will decrease with increasing distance travelled (and travel costs), other things remaining equal and thus provides an indirect measure of net willingness-to-pay (i.e. consumer surplus). In the Hedonic Price method the environmental good is assumed to be one of several characteristics that affects property price, e.g. noise, air and water quality and aesthetic landscapes (including river views) and is of little relevance for valuing inland fish stocks.

Stated Preference methods value the environmental good in question by constructing a hypothetical market for the good and this is the major criticism of the approach. However, stated preference methods are useful because they provide a mechanism for estimating both use and non-use value of a future change in environmental goods. Stated Preference methods can be divided into direct and indirect approaches. The direct Contingent Valuation (CV) method is the most commonly used, but mostly for recreational fisheries. Amongst the papers presented to LARS2, only Alam (2003) attempted to value commercial fisheries based on this approach, during a study in Bangladesh. Over the past few years indirect approaches of Contingent Ranking (CR) and Choice Experiments (CE) have also gained popularity. The main difference between these two approaches is that while the CR method typically is a two-options (referendum) approach, CE employs a series of questions with more than two options that are designed to elicit responses allowing for estimation of preferences over attributes of an environmental state.

A Contingent Valuation (CV) survey constructs scenarios that offer different possible future government actions. Under the simplest and most commonly used CV question format (binary discrete choice or closed-ended method), the respondent is offered a choice between an action that maintains the status quo policy and one having a greater cost (e.g. increased taxes, higher prices associated with regulation, or user fees). Basically the respondent provides an in favour/not in favour answer with respect to the alternative policy (versus the status quo). Factors such as what the alternative policy will provide, how it will be provided and how much it will cost and how it will be charged for (i.e. payment vehicle), are specified.

An alternative elicitation method is open-ended questions where respondents are asked directly about either: (a) how much they are willing to pay (WTP) for a service or the increase they are willing to pay to maintain access to that service; or (b) how much they are willing to accept (WTA) as 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). The choice of WTP or WTA depends on assumptions about entitlements and whether the change is an improvement or deterioration in environmental quality. Generally, WTA is only used where there are clear property rights to the *status quo* and changes are 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, Jones and Brown 1994; Riechers and Fedler 1996; Weithman 1999; Navrud 2001 for reviews) and of part-time or artisanal commercial or subsistence fisheries, which are comparable to "leisure" activities. In addition, the value that nonusers place on fisheries has to be considered if total economic value of the fishery is to be evaluated.

Whilst the benefits of using CV methods to value resources where no direct market value is available, such as maintaining a pristine habitat or conserving species with no economic value, are important for influencing politicians and decision makers, the methods are also open to criticism. For example, individuals with pro-environmental tendencies are willing to pay more than the general public, thus increasing the WTP estimate (e.g. Kotchen and Reiling 2000). Similarly, when financial outlay becomes a reality, individuals tend to be less willing to pay than when it is only a query. They are also difficult to use in developing country situations where rural people have no perception of the economic value of environmental goods and services.

These WTP or consumer surplus estimates can, however, be used in benefit-cost analysis to evaluate the 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 and demonstrated that the benefits to anglers alone outweighed the costs of low-flow alleviation programmes in two of seven rivers evaluated. The value to other recreational and non-users 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 be also net losses associated with a change in management regimes, which benefit recreation including fishing, but constrain commercial enterprises such as hydropower generation.

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 fishers experience certain value components 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 subsidised. Net economic value of commercial fishing comprises consumer surplus 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 when comparing 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 are consumer surplus 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).

Expenditures by anglers or commercial fishers represent revenues and jobs generated in local economies. There are three types of economic impacts: (1) direct impacts, which are the purchases made by fishers, including travel, accommodation and food costs; (2) indirect impacts, which are the purchases made by businesses to produce goods or services demanded by fishers; and (3) 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. This can add considerable value to the fishery activities. For example, an impact analysis on fishing expenditure (US$25.6 million in 1990) on the regional economy bordering Lake Texoma (USA) found the direct, indirect and induced impacts of this expenditure was directly associated with US$57.4 million in total business sales, US$23.3 million in value added and 718 jobs (Schorr *et al*. 1995).

SOCIAL BENEFITS

Four categories of social value relate to river 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).

Cultural values represent a collective feeling toward fishes and fishing. Fishing in rivers 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). Data on the incidence of human illness can be obtained from the local health office or hospital.

Social impacts are very elusive (Vanderpool 1987). They relate to quality of life and social wellbeing caused by fishing (Gregory 1987), including improvements in rural livelihoods (Bene and Neiland 2003; Hand 2003). For example, attracting lots of recreational anglers to a river 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 river fisheries are, typically, difficult to quantify (Kearney 1999, 2002; Table 2). Because most rivers are impaired in some way, there is an increasing trend towards intervening either to improve the functioning of degraded systems or to restore them (Cowx 1994; Cowx and Welcomme 1998). Thus, much river fisheries management aims to mitigate or rehabilitate the adverse human-induced changes by manipulating the ecosystems in an attempt to gain positive benefits (Brown 2003). 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 fishery users 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. Indirectly, in some northern temperate countries, fishery stakeholders, especially recreational fishing societies, have pushed governments to formulate environmental legislation and were the driving forces behind improvements to river quality.

However, not all measures adopted under traditional inland fisheries management are considered positive. For example, common management measures such as stocking and introductions (Araujo-Lima *et al.* 2003) are serious threats 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, Hickley and Gledhill 2002). This awareness of environmental issues and diversity of ecosystems by fishery protagonists (e.g. Kearney 1999; Connelly, Brown and Knuth 2000) is paramount for ecosystem based management (e.g. Olsson and Folke 2001) and sustainability, assuming that ecological responsibility is achieved. Furthermore, indigenous knowledge of the fishing communities and informal (local) institutions can play an important role in the sustainable management of fishery resources (e.g. Mackinson and Nøttestad 1998; Berkes, Colding and Folke 2000; Johannes, Freeman and Hamilton 2000).

A NOTE ON THE VALUATION OF BIODIVERSITY

It remains the exception for values to be put on diversity in aquatic resource management planning. Some part of this may be due to difficulties in understanding the concept of diversity while another factor may be the quite considerable difficulties in collecting and analysing the required information. Diversity, however, underpins our existence on this planet; it should not be ignored. When it is not, decisions that might otherwise by made solely on political grounds, should be further refined by an economic and ecological examination of the issue.

Biodiversity is a concept that describes the way in which the different goods (or components) and services (or functions) of an ecosystem are organised. It has three parts - genetic diversity, species diversity and ecosystem diversity. Essentially, for all three parts, it is the degree of variety in the natural resources - measures of the richness and distribution within the system; it should not be confused with the biological resources themselves. For example, genetic diversity describes the variation within a particular pool - the number of genes and their distribution, not the pool itself or the characteristics of individual genes. Likewise, species diversity is a measure of species richness and their distribution, but it is not a description of individual organisms. The diversity of ecosystems indicates the number and range of the types of ecosystems that exist in a given area but does not describe the ecosystems themselves. Thus, the valuation of biological diversity is not to be equated with the valuation of resources, although the two sets of values are closely related.

As do biological resources, biological diversity is recognised to have direct and indirect use values. The essence of the difference in the measurement of the value of resources and the value of diversity is that whereas in the former the analyst is concerned with the identification of the gross use values, in the measurement of diversity values, attention is directed towards measuring marginal changes in output that result from marginal changes in relevant factor inputs. Measuring the change in economic activity that results from a specified decline in diversity is one way of estimating the direct and indirect use values of diversity. For example, in the measurement of the direct use values of the species diversity of a coral reef, changes in both the diversity of coral species and the gross amount of coral cover affect fish biomass. The resulting elasticities could be used in calculating the value of coral diversity by estimating the change in the revenues earned from fishing on the reef. An example of a direct use value of ecosystem diversity is the tourist revenues derived from the viewing of coral reefs.

In the case of direct use values, ecological substitutes are often more elusive than economic ones. For example, if favoured firewood for smoking fish becomes unobtainable because of over exploitation but the substitute, which is readily available, is almost as satisfactory in terms of heat output, smoking quality and ease of collection, the economic costs of switching to the alternative will be small. It follows that in this case the direct value of species diversity will be small, although the ecological cost will be high. By contrast, if a favoured firewood can only be replaced by people who collect it having to travel much longer distances, then the economic costs will be high, making also high the direct value of species diversity.

Indirect use values of biodiversity may also have economic or ecological substitutes. Again the benefits gained from degrading a diverse range of valued environmental services should be weighed against the availability and usefulness of substitutes, e.g. the value of marginal wetland may be high because the relatively low availability of ecological substitutes and the high costs of economic substitutes (e.g. water purification plants, water transport, relocation of fishing communities dependent on the wetlands).

In the calculation of direct and indirect use values of diversity, there is room for double counting of resource use values and diversity values and for tradeoffs. Care has to be taken to ensure that diversity values are used separately from resources values to evaluate the impacts on biodiversity of current pressures or threats.

ROLE OF VALUATION IN RIVER FISHERIES MANAGEMENT

The value of maintaining and supporting river ecosystem function is illustrated in Figure 3. A healthy ecosystem generates wealth for the local and regional economies, which implicitly supports rural livelihoods. In this cycle, fisheries plays three important roles:

- As a public good for all to use and enjoy;
- Generating revenue for local economies;
- As a catalyst for ecosystem regeneration and community engagement.

These three elements must be interdependent to succeed. Without the landscape, biodiversity and aquatic resource value people are not drawn to use rivers. These qualities depend on good management and environmental control of the entire river ecosystem, including the river catchment and its biodiversity. If the river corridor does not attract participation in resource use there is no catalyst for economic development or stimulus for community engagement and there is no reason to maintain or enhance the river environment.

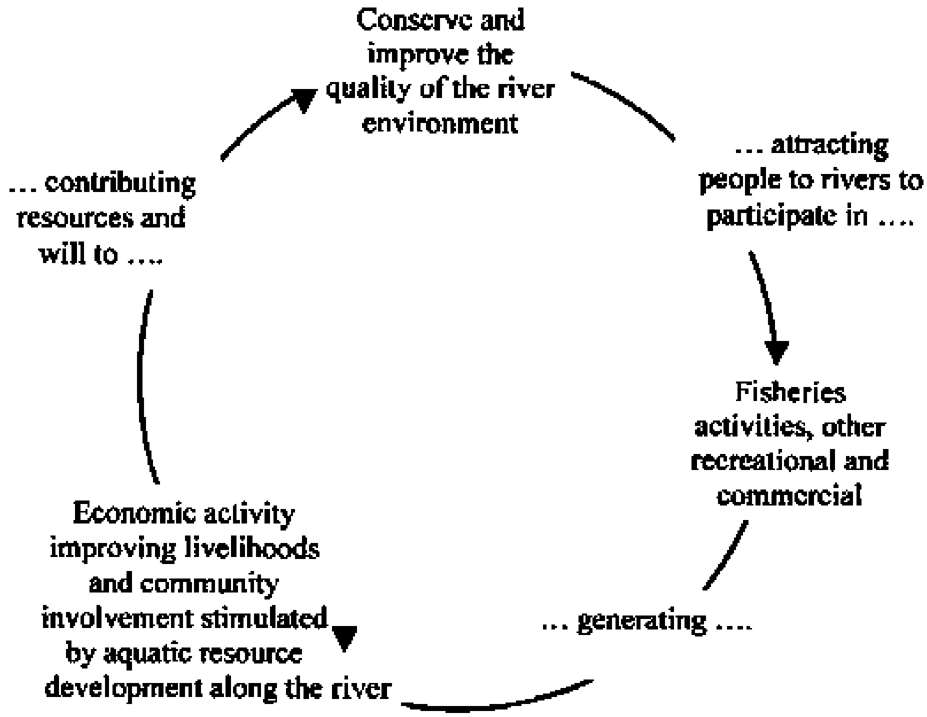


Figure 3. The river cycle illustrating the importance of participation in maintaining ecosystem function and supporting rural livelihoods.

The underlying tenet of these arguments is to recognise the value of river fisheries and the ecosystem for sustaining rural livelihoods. This is highlighted in DFID's sustainable livelihoods framework (Figure 4) that shows how the values of various assets are the focal point for influencing policy to improve the wellbeing of riparian communities. The livelihoods approach addresses issues related to vulnerability and reduction of risk associated with resource exploitation patterns. These are values that are difficult to assess but low risk and reduced vulnerability to poverty are very important features of sustainable livelihood strategies and maintaining food security.

Unfortunately, river fisheries are threatened by a wide array of factors and therefore communities are highly vulnerable to change, but anthropogenic disturbance seems to underlie the decline and extinction of many fish species (see Cowx, 2002b for review). The main perturbations can be broken down into five key problems, viz: species introductions and translocations, impoundment of rivers (dams and weirs, water abstraction and water transfer schemes), water quality deterioration (pollution, eutrophication, acidification), habitat degradation and fragmentation (channelisation and land use change, mineral extraction) and overexploitation. These problems seem to be universal. Although many of the issues are being addressed in developed countries through environmental legislation, the rate of progress in reversing the impacts is pathetically slow. Furthermore, the cost of implementing rehabilitation programmes or seeking alternative solutions to the demands on water resources, which underlies many of the issues, is prohibitive and at best only a *status quo* is being achieved with respect to habitat quality and at worst, as is still commonly found throughout the developing world where financial resources are limited, progressive deterioration is rife.

One of the key reasons for the problems facing fisheries is that the value of the fisheries resource is usually ill defined and poorly represented from an economic and social perspective (Cowx 2002a and 2002b). Fisheries are traditionally managed based on the quality of the fishing experience or volume of catch and few are managed from an economic perspective (Cowx 2002a), an issue born out by the paucity of information on the economic value of fisheries (e.g. Baker and Pierce 1997; Peirson *et al.* 2001). A large number of recent works underline the high potential of small scale fishing activities for economic development (both at local and national levels) but systematically highlight how poorly the true (economic) value of this sector is reflected in official statistics and discussions of food security and livelihoods (e.g. European Commission 2000; Kaczynski and Looney 2000; Anon 2001). As a consequence, fish and fisheries are generally not considered of sufficiently high priority or value and thus suffer in the face of economically and socially higher priorities, e.g. agriculture, hydroelectric power production or other water sports. It is also usually presented as the main constraint for the design of appropriate policy for aquatic resource management, both at the national and regional levels. If fisheries are to be promoted in the future, there is an urgent need to provide robust, defensible, social and economic valuation of aquatic biodiversity and fisheries (Cowx 2002a). Once this information is available, value will be a powerful tool for arguing the case of fisheries. However, it must be recognised that it is not the only tool to be used because the economic value of, for example, a major water resource scheme may far outweigh fisheries value. This is primarily because the methods used for valuation are often fisheries specific and do not consider the upstream economic value in terms of aesthetic and conservation value and the provision of goods and services, or the downstream value associated with the service sectors. To reverse these philosophies is going to be a major challenge to fisheries and conservation managers, but neither will be achieved if the true economic value of preserving fisheries is not enunciated (Cowx 2002b). As mentioned earlier, accurate valuation of the fisheries should be a major thrust of fisheries development activities in the immediate future.

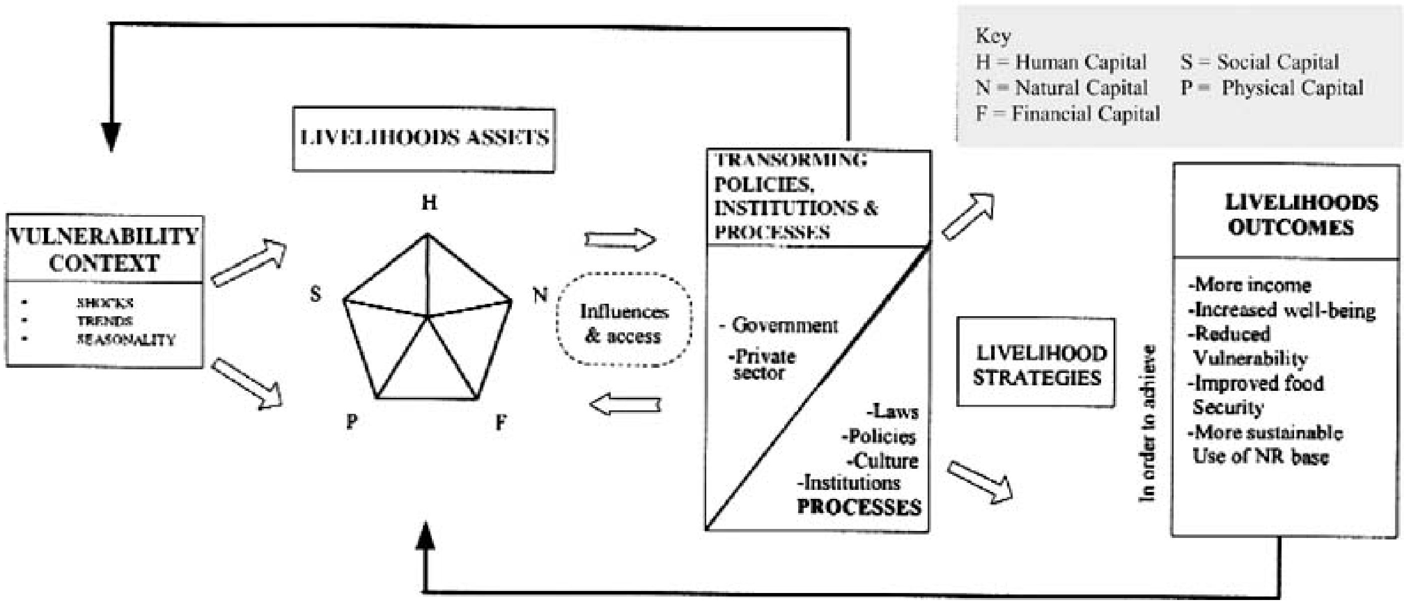


Figure 4. DFID sustainable livelihoods framework

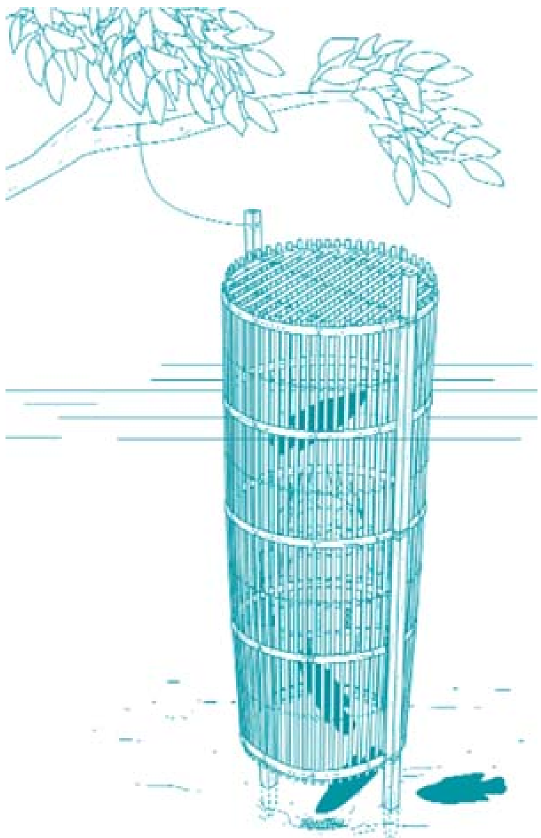
In the past, management of fisheries resources has been based on interpreting information on the fish stocks and reacting to shifts in availability (Cowx 1996). Integral within this approach are adequate stock assessment procedures that provide the baseline information on which to manage the fisheries resources. However, when reviewing the problems relating to river fisheries it is clear that this approach is inadequate. Increasing pressures on aquatic resources dictate that fisheries exploitation and conservation can no longer be treated in isolation and an integrated approach to aquatic resource management is required (Cowx 1998). Similarly, fish biodiversity is being constantly eroded, not only by exploitation of fish directly but mainly through degradation of their habitat. Fortunately, the demands for sustainability that grew out of the World Summit on Sustainable Development (WSSD) in 1992 have put emphasis on the need not only to manage exploited resources but also promote biodiversity. Unfortunately the WSSD did not endorse fisheries, but this was rectified in the 2002 WSSD in Johannesburg. Consequently, conflicts between various user interests must be resolved by involving all stakeholders in the management process and defining priority areas for conservation and preservation of bio-diversity (Brummett and Teugels 2003; Darwall and Vié 2003). This can be achieved through integrated aquatic resource planning and management. River basin management plans, at both the national and multi-national scale, which for example will be obligatory under the new European Union Water Framework Directive, will support this process, but the profile of fisheries exploitation in the widest sense and fish conservation need to be raised and be better integrated into the planning process. Without this involvement the future of river fish and fisheries remains uncertain.

Aquatic resource planning and management, as suggested above, must be a multi-disciplinary, interactive approach dealing with all the existing and potential user groups, including adjacent land use. It should allow wider issues than those related to a single activity, in this case river fisheries, to be taken into account during the process of decision-making about an activity and its likely effect upon the environment and other activities, or conversely the likely affect of other activities on fisheries. For this process to be effective, data on the social and economic importance of each resource are needed, without it, economically strong activities such as hydropower development will override.

Many of the sources of conflicts between aquatic resource users lie in the difficulties of communication between user groups, the lack of a mechanism for dialogue or in a failure to understand common objectives. However, the failure of dialogue between user groups frequently arises through a lack of willingness on the part of the stronger group to discuss resource allocation with the minority group. One solution might be the better co-operation of players within, for example, a co-management framework (Sen and Raakjaer Nielsen 1996), challenging the present routines. An essential element in co-management is continuous shared responsibility and decision-making between government, fishers and other stakeholders. Co-management is one possible mechanism that could ensure that the human element is accounted for. Inclusion of all stakeholders in co-management systems ensures that decisions better reflect local, social, economic and environmental conditions. In developing countries co-management is being promoted on many fisheries with the devolvement of responsibility to the riparian communities. In Europe and North America, less participatory co-management approaches are likely to apply, with many inland fisheries being jointly managed by fishers and government officials. In both cases, however, the objective remains the same, sustainability of the exploitable resources and biodiversity for future generations.

Similarly, there is a need to develop partnerships with stakeholders in affected ecosystems to strengthen and implement fish and fisheries related activities and develop mechanisms to influence other players. To achieve this, scientists must expand their range of activities from monitoring and reporting the status of stocks and species to more influential and preventative work. They must use the best available data to educate other stakeholders and the wider public. They need to be involved in accurate environmental impact assessments and rehabilitation programmes to argue the case for fish and fisheries, i.e. there is a need to develop a risk based approach to fisheries management. There is also a need to develop fiscal measures, such as the 'polluter-pays principle' and enforce legislation through the appropriate channels and institutions. This will only be achieved through valuation of fisheries resources, an issue that is acting against the fisheries lobby and will be essential for integration into river basin management plans. As previously stressed, there is an urgent need to adapt environmental economic evaluation tools to value the social and economic importance of freshwater fisheries and biodiversity. Until this is undertaken fish and fisheries will continue to be given low priority in any consultation process and it will remain difficult to attract investment or credit for protection of the fisheries.

Irrespective of the mechanism of implementation, the managers and resource users need a true economic value of their resources to defend their position in conflict and development scenarios. This will increase the capacity of beneficiaries of river (inland) fisheries to communicate and influence at all levels of society. In this context it is important that information on values is conveyed to politicians, planners and stakeholders in simple language. It is therefore important to understand how value is interpreted within communities, i.e. between rich and poor and fisher and non-fisher. This can only be achieved if stakeholders in the fishery sector understand the motives, modes of operation and reward systems of other spheres of society and engage in cooperative interchange. More effective management of fisheries resources also requires scientists to learn new skills to interact in complex, disorderly and confusing arenas and not producing scientific information in the vain hope that managers and policy makers will use it. Finally, the methods to value aquatic resources and environmental goods and services do exist. The lack in progress in this field is largely because the fisheries scientists and managers do not have active dialogue with experts in environmental and ecological economics. This needs to be promoted in the drive towards sustainable use of aquatic resources in general and river fisheries in particular.



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