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J. S. Wallace, M. C. Acreman and C. A. Sullivan

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The sharing of water between society and ecosystems: from conflict to catchment-based co-management

J. S. Wallace*, **M. C. Acreman and C. A. Sullivan**

Centre for Ecology and Hydrology, Wallingford, Oxfordshire OX10 8BB, UK

Human uses of freshwater resources are increasing rapidly as the world population rises. As this happens, less water is left to support aquatic and associated ecosystems. To minimize future human water shortages and undesirable environmental impacts, more equitable sharing of water resources between society and nature is required. This will require physical quantities and social values to be placed on both human and aquatic ecosystem requirements. Current water valuation systems are dominated by economic values and this paper illustrates new quantification and valuation methods that take more account of human well-being and environmental impacts.

The key to the effective implementation of these more equitable water allocation methods is the use of catchment-based integrated water resources management. This holistic framework makes it possible for human and ecosystem water requirements and the interactions between them to be better understood. This knowledge provides the foundation for incorporating relevant social factors so that water policies and laws can be developed to make best use of limited water resources. Catchment-based co-management can therefore help to ensure more effective sharing of water between people and nature.

Keywords: water scarcity; equitable allocation; integrated water resources management

1. INTRODUCTION

Until comparatively recently, there has been little debate about sharing water between society and nature. This is not surprising since there is so much water on the planet; we know that two-thirds of the Earth's surface is covered with oceans and textbooks on water show that the total volume of water on the Earth is vast: *ca.* 1.4 billion (10^9) km^3 (Maidment 1992). However, less than 1% of this water is fresh and accessible, the rest being either saline, frozen or in deep underground aquifers. This still very large (11 million km^3) amount of freshwater is distributed very unevenly around the globe and this is reflected in the geographical variation in vegetation and human population. Recent attempts to map this variation of freshwater resources have not only quantified the variation in freshwater resources around the world, but also how human demand for these resources varies globally (Shiklomanov 1991; Alcamo *et al.* 1997). The combination of estimates of water supply with demands for water has revealed several very important issues. Firstly, although future water supply may be influenced by changes in amounts and timing of local rainfall owing to climate change and/or variation, it is the demand for water for domestic, industrial and agricultural uses that will dominate future water scarcity. Furthermore, of these three human uses of water, irrigated agriculture accounts for by far the greatest amount (between 65% and 75% of total water use).

The global consequences of this have been illustrated by several authors (Fischer & Heilig 1997; Falkenmark 1997; Gleick 2000; Vorosmarty *et al.* 2000) and summarized by Wallace (2000) using figure 1*a*,*b*. This indicates that the most acute water shortages are currently in North Africa, where the *per capita* annual renewable freshwater resource is less than *ca.*1000 m^3 and insufficient to meet human needs (for growing food, for domestic purposes and a *per capita* industrial allowance). Falkenmark (1997) and others (Fischer & Heilig 1997) recognize that there can be human water requirements in excess of 1000 m^3 (e.g. in areas where agriculture is largely irrigated and/or where domestic and industrial requirements are high) and conclude that there could still be some water scarcity when *per capita* annual renewable freshwater resources are between 1000 and 2000 m^3. Figure 1*a* shows that some degree of scarcity currently exists throughout Southern Africa and the Middle East and that by 2050 a staggering 67% of the world's population (6.5 billion people) may experience some water scarcity (figure 1*b*). Furthermore, as many as one in six people (1.5 billion) may have insufficient water to meet their requirements for growing food and domestic water. The main issues in these water-scarce areas therefore tend to focus on the human problems of alleviating hunger and poverty and water supply for household use and sanitation.

At the same time as evidence for this immense human water crisis has emerged, there has been growing concern for the impacts of human appropriation of water on ecosystems. Because humans use water from rivers, aquifers

* Corresponding author (jimwallace@easynet.co.uk).

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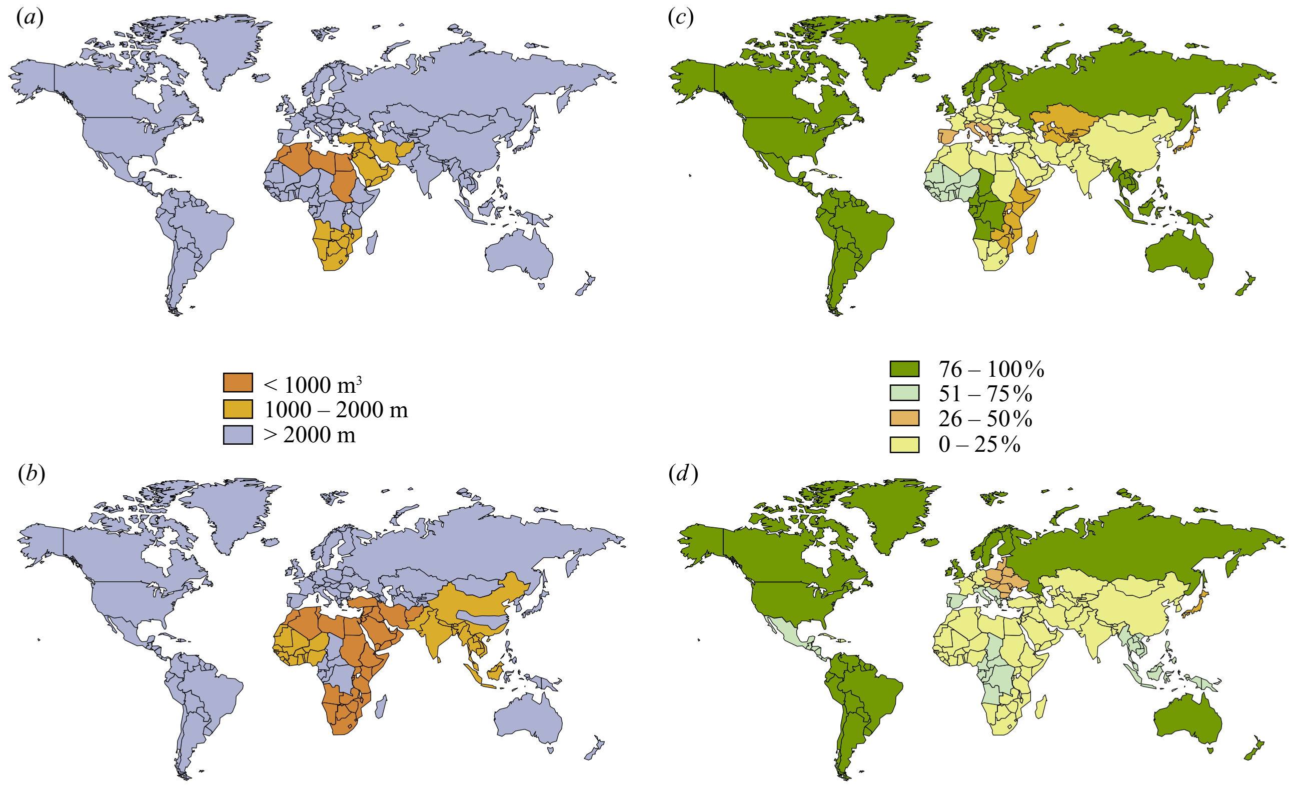

Figure 1. *Per capita* annual renewable freshwater (*a*) now and (*b*) in 2050, and the percentage of this left for aquatic ecosystems after meeting all human needs (*c*) now and (*d*) in 2050.

and lakes (the 'blue water' system), the ecosystems primarily affected by human water use are aquatic and riparian zone ecosystems. In areas where water is less scarce, environmental concerns often dominate and the main issues tend to be with water quality, environmental impacts and ecosystem protection. However, environmental issues are also becoming increasingly important in water-scarce areas. For example, a key issue identified by the Global Water Partnership was the increasing competition for water resources between the agricultural and environmental sectors (see Rijsberman & Molden 2001). Rijsberman also points out that several studies have indicated that while the environmental sustainability lobby are advocating a decrease in water allocation to irrigated agriculture (e.g. by 8%; see Alcamo *et al.* 2000), the agricultural communities are arguing for an increase in water allocation to meet future food requirements (e.g. by 17%; see Rijsberman & Molden 2001).

Figure 1*c*,*d* gives a broad global picture of where the pressures on aquatic ecosystems are and may be greatest, now and in 2050. The figure shows what percentage of the total annual renewable freshwater resource would be left in a scenario where all human needs were met. This is simply calculated by subtracting 2000 m^3 $person^{-1}$ yr^{-1} from the total annual renewable freshwater. Figure 1*c* shows that, at present, around one-third of the land area of the world would have less than 50% of the available water resource to support aquatic ecosystems and most of this area would have less than 25%, a figure below which severe environmental impacts may occur. Even when the water available for aquatic ecosystems is greater than 50% of the total resource there may be important environmental impacts. For example, experts have suggested that in Australia the probability of having a healthy river falls from high to moderate when the hydrological flow regime is less than two-thirds of its natural condition (Jones 2002). The areas where there is least water for aquatic ecosystems are in East and West Europe, Northern and Southern Africa and East, West and South-Central Asia. One area, Northern Africa, currently has insufficient water to meet even the lower level of human requirement (i.e. 1000 m^3 $person^{-1}$ yr^{-1}), and this may also be the case for Western and South-Central Asia by 2050. Therefore, unless food is imported and/or production intensified in both rain-fed and irrigated agriculture, there may be little or no water left to support ecosystems in these regions. Clearly, figures based on annual average volumes mask important aspects of the duration, frequency and timing of water availability for aquatic ecosystems. This is discussed in more detail in § 3.

By 2050, the global situation under the above scenario will have changed markedly, with over 40% of the land area having less than 25% of the available water resource to support aquatic ecosystems (figure 1*d*). The number of areas where there is a danger of severe environmental degradation increases by 2050 to include Western Africa, Eastern Africa and Central Asia. By this time, large areas (over one-third) of tropical forest and wetlands will have less than three-quarters of the available water resource and nearly 20% are in areas where there is less than one-quarter of the water resource left after meeting all human requirements. The crude analyses illustrated in figure 1 are only intended to be indicative, since they have clear limitations, mainly owing to their spatial (regional average) and temporal scale (annual average), gross assumptions about *per capita* water requirements and

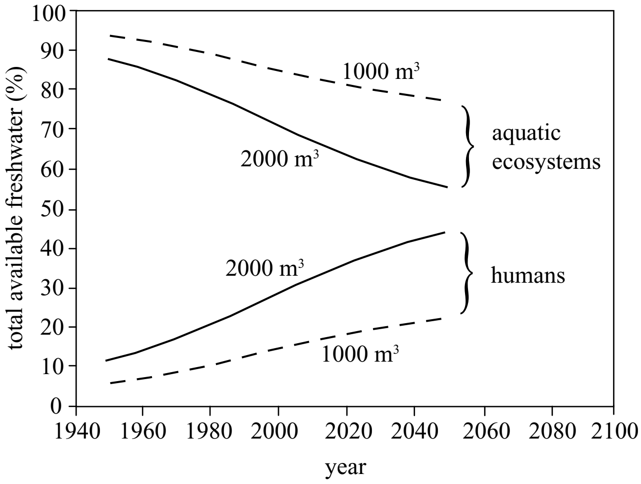

Figure 2. Change in human water requirements and aquatic ecosystem water availability with time, assuming two levels of human use: 2000 m^3 $person^{-1}$ yr^{-1} (continuous lines) and 1000 m^3 $person^{-1}$ yr^{-1} (dashed lines).

further assumptions about the proportion of total food production from irrigated agriculture. Furthermore, the analysis assumes that human water uses are consumptive, whereas in reality some of this water may be able to sustain further human needs and/or support ecosystems downstream. This is discussed further in § 4.

Despite the shortcomings of the simple picture presented here (figure 1), it remains sufficient for making three important points, as follows:

(i) future global water scarcity will affect most of the world's population;
(ii) population growth dominates future water scarcity; and
(iii) there are widespread implications for aquatic ecosystems in many parts of the world.

More accurate water scarcity assessments, with higher time and space resolutions, are described in Wallace & Gregory (2002). However, these are all focused on human water requirements and little or no attention has been paid to the global assessment of the impact of human use of water on aquatic ecosystems. An assessment of this type is being compiled by Smakhtin *et al.* (2003). Their approach involves using a global hydrological model with 0.5° spatial resolution and relating calculated river flow variability to indices of ecosystem integrity. They estimate environmental water requirements to range from 20% to 50% of the total water available and produce global maps that show the regions of the world where current human water use is already in conflict with environmental water requirements. There is broad agreement between these regions and those indicated in figure 1*c*, although the higher-resolution analyses by Smakhtin *et al.* (2003) show areas of conflict within nations and continents that our regional analysis does not reveal.

The crux of the dilemma facing mankind is illustrated in figure 2. This shows how the human and aquatic ecosystem proportions of the total freshwater available on the planet would change between 1950 to 2050 under the scenarios where human needs were met at either 1000 or 2000 m^3 $person^{-1}$ yr^{-1}. In this very short period of 100 years, the balance of water used by humans and left for nature changes drastically. Human requirements in the 1950s would have amounted to *ca.* 10% of the available resource, leaving the rest for nature. At present, to avoid all water shortage, humans would require *ca.* 30% of the available resource, but this could rise towards 50% if populations grow according to the current UN projections and every person used 2000 m^3 yr^{-1}. In this scenario, aquatic ecosystems would be left with around half of the available freshwater. With 40% of the entire land surface and 20% of tropical forest and wetlands having access to less than one-quarter of the freshwater resources, the environmental consequences would be enormous. Alternatively, by 2050, 6.5 billion people may live with water scarcity, and 1.5 billion with inadequate water for their food and domestic needs. By comparison, in 1950, this simple analysis indicates that only 50 million people lived under water scarcity and relatively few had inadequate water for their food and domestic requirements.

People clearly benefit from the direct use of water for domestic purposes, agriculture and industry, but they also benefit from the water provided to aquatic ecosystems. Ecological processes keep the planet fit for life, providing non-agricultural foods, air to breathe, medicines and much of what we call 'quality of life' (Acreman 1998). In particular, millions of people worldwide, particularly in poor communities, depend greatly on natural resources of aquatic ecosystems, including fish and timber. In many communities who live in or near wetlands, their social structure is adapted to the hydrological regimes, such as the annual flood (Acreman *et al.* 2000). Many people benefit from hydrological functions of aquatic ecosystems, e.g. river wetlands that can provide flood protection and water quality improvement of floodplains and marshes (Hewlett & Hibbert 1967). In addition, there are widely held views that the human race has a moral duty to protect the biodiversity of our planet (Acreman 2001). Thus, providing water to aquatic ecosystems also serves economic, social, ecological and ethical needs.

Clearly, water links society and nature, so the remainder of this paper explores societal and ecosystems water needs and how this precious resource can be shared between them. The main focus of this paper is on water quantity, as we do not have the scope to address water quality issues in detail. Is the picture quite as bleak as it appears above, or can we accommodate future human requirements for water without unacceptable impacts on aquatic and terrestrial ecosystems? What can catchment-based water management techniques contribute to the resolution of this apparently intractable dilemma?

2. SOCIETAL WATER NEEDS

Different sectors of society use water in a variety of ways. Wide variations also exist in the relative importance of water in different countries, and therefore assessing how societies benefit from water use is very complex. In addition to economic criteria for assessing the importance of water to society, other important criteria include human health, aesthetic and spiritual values, as well as some recognition of the intrinsic psychological and empowering value of simply having secure access to a convenient water supply. It is interesting to note that while domestic water needs are absolutely fundamental to our survival, emphasis placed upon them within many water management strategies is rather low. Water allocations continue to be heavily weighted in favour of agriculture, with often inadequate amounts being provided for domestic use. Water management can therefore be improved by quantifying all of the ways in which water use benefits society. In economic terms, this is referred to as an investigation of the 'returns to water', but it should be stressed that this is only one way in which the benefits of water use can be assessed.

(a) *The returns from water use*

There are many issues to be considered in any quantification of the returns arising from the use of water, and these must not be identified using only economic criteria. When decisions are made about how water is managed, economic and political considerations are often given priority. For example, when water storage is provided for both industry and agriculture, the justification for the associated expenditure is provided by an examination of the economic returns on the capital invested. These would be assessed on the basis of electricity generation potential or increased crop outputs, and often little else is considered. However, when assessing domestic water provision, a problem is encountered in defining appropriate measures of the returns on investment. Although monetary measures are appropriate for many criteria, certain important attributes of the value of domestic water cannot be measured in this way. For example, no definitive method of valuation has been agreed to quantify effectively the value of good health, or the lower child mortality rates normally associated with clean domestic water provision. Similarly, the difficulties associated with valuation of environmental attributes are well known. While some progress has been made on the valuation of a range of natural resources (Abramovitz 1997; Roodman 1999; Sullivan 2002), methods to value ecosystem functions and services are less developed, despite the ever-growing literature on this subject.

Water is increasingly perceived as a 'strategic resource' and water accounts have now started to be constructed that stress the economic importance of water. Although these water accounts can help in identifying problems related to the emission of pollutants and to the general management of water, they are considerably limited in not recognizing the real global importance of water. For example, social and/or political considerations are not included, although they are very important in the management of water in developing countries. Neither should the management of water be limited to issues such as waste management and land use. All of the above social, political, economic and environmental considerations are very closely linked and should be treated as such if the management of water resources is to improve. The concept of 'critical natural capital' (Berkes & Folke 1994; Noel & O'Connor 1998) recognizes the importance of ecosystems and stresses the idea that, for instance, some key species may be critical to an ecosystem's function, or that the pollution of a certain habitat can have repercussions on other habitats and species. The controversial work of Costanza *et al.* (1997*a*) has served to highlight the importance of ecosystem functions and services. However, it is still unclear how these essential attributes of nature can be incorporated into human management systems. As a result, it is extremely difficult for market-driven water managers to incorporate the values of nature into day-to-day water management techniques.

If we look at the returns from water use by sector, we can see that in some countries, different sectors play different roles in economic performance, and the use of water in these sectors may be a determining factor in the economic progress. The relationships between water availability, sectoral water uses, GDP and the human development index can be examined using simple linear correlations, as shown in table 1. For example, in the agriculture sector in poor countries there is some positive correlation between water use and the contribution of agricultural outputs to the GDP, whereas in richer countries, there appears to be no clear relationship. At first sight, this suggests that agricultural water management in the poorer countries may be more economically important than in richer ones. However, these figures take no account of rain-fed farming systems that provide significant amounts of food throughout the world, much of which may be unaccounted for in national accounts owing to home consumption.

Table 1 also shows a similar picture in the industrial sector. In poor countries, higher levels of industrial water use tend to generate greater contributions to the GDP, whereas there is no such correlation in rich countries. This suggests that in poor countries, when water is used for industry, the economic returns are of greater importance to the nation than in richer countries. In poor countries, there is a high negative correlation between the contribution of agricultural activity to the economy and the country's score on the HDI (UNDP 2001), indicating that a high level of agricultural dependence may be associated with lower levels of economic and social well-being. This also confirms that developing economies heavily dependent on agriculture have little influence on world commodity prices, often getting very little return for the use of their resources, including water. From this sample of countries, there also appears to be a stronger link in the higher-income group between *per capita* water use and *per capita* GDP. This again suggests that richer countries are in a better position to get higher benefits from their water use than poor countries. This is clearly an area of research that should be explored much further, since it suggests that how water is managed within an economy can have a direct impact on the economic welfare of society. A related issue, which merits further work, is the distributional impact (in society) of the ways in which water is managed, and how more effective water management could contribute to poverty alleviation. Examples of poor distributional impact can be seen in many of the large dams throughout the world that have been built for commercial irrigation schemes or electricity generation. In many of these schemes the beneficiaries tend to be the richer members of society and/or the political elite, while those coping with the impacts of the scheme are often poor subsistence householders who not only do not have electricity, but are also likely to lose important livelihood benefits provided by functioning downstream ecosystems.

Table 1. Correlations between economic performance and sectoral water use in selected countries.
(Note: in the higher GDP countries, no country has less than 0.1 km^3 *per capita*, whereas in the low income group, all of the countries have less than 0.1 km^3 freshwater withdrawals. Data sources: population: World Resources Institute (2000) tables HD.1 and SCI.1 and UNDP (2001); water resources: World Resources Institute (2000) table FW.1, and Gleick (2000); GDP and sectoral contributions: UNDP (2001).)

	correlation between				
	agricultural water use (%) and contribution to value added to GDP from agriculture	industrial water use (%) and contribution to value added to GDP from industry	value added to GDP from agriculture and HDI score	water withdrawals and HDI value	water withdrawals and GDP *per capita* value
countries with high GDP *per capita*[a]	−0.03	−0.07	−0.06	0.51	0.47
countries with low GDP *per capita*[b]	0.21	0.69	−0.51	0.11	0.27

[a] Countries with *per capita* GDP of over US$10 000, and *per capita* water withdrawals of less than 0.5 $km^3\ yr^{-1}$ (Barbados, Cyprus, Czech Republic, Denmark, Ireland, Kuwait, Netherlands, Norway, Singapore, Slovakia, Slovenia and Sweden).
[b] Countries with *per capita* GDP of less than US$10 000, and *per capita* water withdrawals of less than 0.1 $km^3\ yr^{-1}$ (Benin, Botswana, Burkina Faso, Burundi, Cambodia, Chad, Côte d'Ivoire, Ethiopia, Eritrea, Ghana, Haiti, Kenya, Lesotho, Malawi, Mozambique, Niger, Nigeria, Tanzania, Togo and Uganda).

(b) *The need for equity in water allocation*

Equity in water management must address how water is shared both between people now, and between current and future generations. Inequity in water allocations today means that there are *ca*. 1–2 billion people who lack access to adequate safe water, while millions of others consume some 14 times their basic minimum requirement for domestic purposes. The reasons for this situation are complex, but undoubtedly they need to be addressed by more equitable and sustainable policies for water management (Biswas 1989). However, when we consider how this situation may be exacerbated in the future, as illustrated by figure 1*a*,*b*, it is clear that significant actions need to be taken soon if we are to avoid greater inequity of this kind in the future. Extreme inequity can lead to human disasters (disease, famine etc.) or even wars, which are not taken into account in figure 1.

The impact of human population growth is also a major factor when considering the future challenges for the equitable allocation of water between humans and aquatic ecosystems. This is illustrated in figure 2, which shows how increases in human water uses over the next 50 years may reduce the amount of water available for aquatic ecosystems. These figures highlight the need for more holistic water management that includes ecological requirements, otherwise irreversible changes in ecosystems may occur, resulting in a significant loss of benefits to mankind. In determining water allocations it is necessary to recognize that ecosystem goods and services are important in different ways to different groups. This is can be illustrated by the ways in which in-stream flows are regarded in different parts of the world. For example, in the Western US many river courses have been totally disrupted (McCully 1996). However, recent experiments on flood releases from reservoirs, such as Glen Canyon Dam on the Colorado, have been undertaken to determine the flows required to maintain channel morphology (sediment transport) and provide habitat for key fish species (Rubin & Topping 2002). In the Murray Darling Basin, in Australia, the maintenance of an effective in-stream ecosystem flow requirement has been the key objective of their water policy. The Australian management approach has been developed to address the ecological crisis resulting from widespread algal blooms, and salinization both of the soil and of estuarine waters, such as at the mouth of the Murray–Darling system. It has required the development of significant institutional agreements, along with inter-state cooperation on trade in water rights (Powell 1998). In the USA, policies in favour of large-scale electricity generation facilities have been developed on the basis of being economically efficient. Although good and bad examples of water management can be found in both the USA and Australia, the examples provided here illustrate the different emphasis that may be placed on ecological issues in different places.

There are other examples throughout the world where some progress has been made to incorporate ecological water needs into policy for water management. Most notably, in South Africa, the Water Act (1998) has been accepted into national planning systems in such a way as to ensure explicitly that an ecological reserve is to be maintained. Even in this case, however, the implementation of this policy is not proving to be easy, as there is insufficient knowledge of the relationships between river flows and ecological health. Another fundamental component of South African Water Law is the commitment to the establishment and maintenance of a basic minimum water allocation to all of the population. This provides an additional limit on abstraction rates, and as a result, legislation has been introduced to regulate organizations involved in any flow-reduction activity, such as commercial forestry. This is a good example of a system that is not only trying to implement integrated water resource management (Falkenmark *et al.* 1999; Falkenmark 2002), but is also trying to incorporate land management. The South African experience also demonstrates that even when

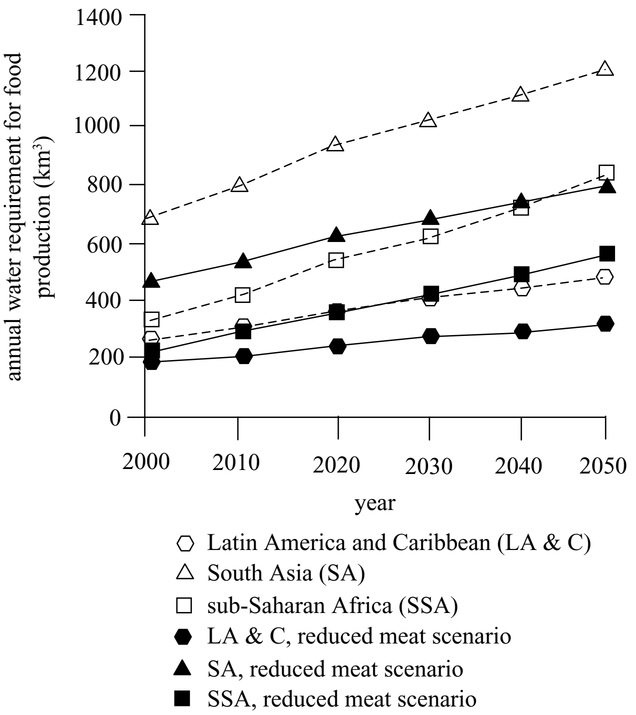

Figure 3. Impact of population growth on water required for food production.

scientific evidence is uncertain, it is prudent to introduce regulations that encourage adaptive management approaches that take account of both human and aquatic ecosystem needs in water-scarce areas.

How land and water are used in agriculture is another factor in determining human pressure on water resource systems. Many of these resources are used to produce non-food lifestyle items, such as tobacco, while at the same time, more resources are being used to produce crops that can generate substantial subsidy payments. For example, market distortions such as these have been widespread as a result of the EU's Common Agricultural Policy, and such policies tend to generate a waste of resources while at the same time having potentially detrimental environmental impacts. This can be demonstrated in the South East of England, where large areas of land are put under extensive cultivation of subsidized crops using high levels of inorganic fertilizers and pesticides. This has created problems of loss of habitats and biodiversity, and an increased risk of groundwater contamination.

All land-use decisions are determined by consumer preferences and farmer crop choice. These decisions can affect water demands and alter the global picture of future water scarcity shown in figures 1 and 2. For example, it has been shown that water consumption rates are lower (by a factor of 5) in the production of protein from vegetable sources than for protein produced from animals (Falkenmark 1997; Penning de Vries *et al.* 1997). Figure 3 provides an illustration of how human water needs could be modified as a result of changing preferences away from animal-based diets. It is generally accepted that as counties develop, people tend to increase the amount of animal protein in their diets, thus increasing water use. While this does conform to the current development paradigm, it is possible that preference for animal protein will change, particularly in the developed world, following concern about intensive farming techniques and animal welfare. Given the significant difference in the level of water requirements associated with different food types, it is possible that changes in dietary patterns could impact significantly on agricultural water demand. Figure 3 provides an illustration of how water needs could be modified as a result of changing preferences away from animal-based diets. This diagram shows how the demand for water resources for food production is likely to increase as a result of demographic change over the next 30 years, and how this pattern is likely to be influenced if dietary choice moves away from animal to vegetable protein.[1] This simplified analysis indicates that there is much potential for changing agricultural water requirements, and changes in the types of food eaten may bring about a significant reduction in the water needed for food production. More detailed analysis is needed fully to investigate the interrelated impacts of population growth and changing consumption patterns inevitably brought about by the normal development process. Clearly, as people in developing countries become richer, their consumption will impact on all forms of production, and on prices, while at the same time changing consumption patterns in more developed economies will also have some effect. Such analysis is beyond the scope of this paper, but there is a need to examine such interactions if we are better to understand future pressures on water resources.

In any economy, all production and consumption involve the exploitation and use of natural resources. It is an ironic fact that this is not reflected in the theoretical foundation on which conventional economic analysis is built. Price and value are determined on the basis of preferences and scarcity (Stiglitz 1979), and at the time when Adam Smith was writing his 'Wealth of Nations' natural resources were considered to be 'so abundant as to be unlimited in supply' (Smith 1786). As a result, market economists build their models of economic processes around a fundamental assumption that nature is a 'free good' (i.e. unpriced), an inaccurate reflection of today's world where both the 'scarcity value of nature' and our preferences to use it, have increased (as shown in figures 1*c*,*d*). The question of how to address these problems more effectively are raised by several economists who are no longer willing to accept the conventional assumptions underlying much of economic theory (Norgaard 1991; Costanza *et al.* 1997*b*; Opschoor 1998).

The value of water varies considerably from place to place and also from use to use. For example, for a given quantity of water, the value derived from its use for navigation may be much less than that from its use in a textile plant. For irrigation, the value of the water may be much higher if the crop grown is carrot than if it were wheat. Crop selection has a big impact both on the use value of water and on the amount of water used. As agriculture has the largest share of water allocation in many countries, the value generated from it needs to be fully quantified and, if necessary, methods introduced to regulate agricultural water use. However, it must be remembered that in many countries decisions about food production are taken much more on the basis of politics than economics. For example, some countries choose to subsidize food production for strategic reasons, to ensure national food self-sufficiency.

One accepted way of valuing water is by comparison with the values that could be generated by its alternative uses. This is known as the 'opportunity cost' of water, and can be used as way of improving water allocation decisions. Unfortunately this is not straightforward, as it is often the case that such calculations suggest that the cultivation of higher-value (non-food) crops would lead to a reallocation of water for that purpose. If this were taken to the extreme then water would be allocated in such a way as to act as a disincentive for food production, causing a problem of food security. In some places such problems have occurred where cash crops such as cotton have been produced in preference to food crops, and when too many farmers made this choice, food shortages have arisen. Nevertheless, the opportunity cost methodology does give some insight into the value of water. There is still much work to do to refine techniques of quantifying any kind of natural capital, including water, and no widely accepted method of valuing water and the ecosystem services associated with it currently exists.

It is now well known that poor groups in society tend to depend more directly on natural resources than do the richer groups (Chambers 1995; Desai 1995; DFID 2002). They are often much more conscious of managing natural resources effectively, and many traditional societies have norms and taboos that serve as a control mechanism on consumption. For example, the Lozi people of the Barotse floodplain in Zambia practise indigenous knowledge systems to manage natural resources of the wetlands (Chiuta 1995). Contrary to popular belief, they are often guardians of a well-maintained environment, although there are also many examples of the poor being the instigators of environmental degradation, owing more to their ever increasing numbers than any deliberate destructive strategy (DFID 2002).

The need to incorporate the important values embodied in nature is a prerequisite to achieving a sustainable future (Faucheux & O'Connor 1998; Daly 1999). There is clearly also a need to consider the range of possible preferences and philosophical positions held by various stakeholder groups when considering the values attributed to nature. In this situation, nature itself is a stakeholder without a voice, and so Jacobs (1997) has suggested that a more eco-centric perspective needs to be taken. One way in which the values of ecosystem goods and services can be incorporated into natural resource management is through the adoption of decision-making strategies, which do not rely exclusively on monetary valuation. Multi-criteria analysis is one example of such a technique, and this approach allows both quantitative and qualitative criteria to be incorporated into a decision, thereby allowing a wider range of values to be incorporated. This is illustrated in figure 4, which shows how riverine ecosystem values can be incorporated into decisions on the management of dams. This figure highlights the wide variety of issues that should be considered when managing water resources. Although there has been some criticism of the application of subjectively defined weights in any multi-criteria technique, such an approach does try to encompass a more holistic view of the complexity of water resources management.

This more holistic approach to water management can also be achieved through the use of integrated water management tools such as the recently developed WPI (Sullivan *et al.* 2002). This approach, which explicitly incorporates environmental attributes into its framework, allows decision makers to discriminate between different locations or communities on the basis of a suite of water-related indices. These indices can be consolidated into five main criteria:

(i) an assessment of water resources, including ground and surface water, variability of those resources, and their quality;
(ii) access (by percentage of population) to water both for domestic use and for irrigation;
(iii) a measure of how water is used for productive purposes;
(iv) values reflecting the capacity to manage water, based on education, health, membership of water user groups and access to finance; and
(v) the environmental impact of water utilization, which currently serves as a proxy for the incorporation of ecological water needs.

Normalized scores (between 0 and 100) for each of these criteria can be identified, largely from existing data; by applying a simple weighted average, it becomes possible to generate an overall index value, WPI, as shown below

$$\mathrm{WPI} = \sum_{i=1}^{N} w_i X_i \Big/ \sum_{i=1}^{N} w_i, \qquad (1.1)$$

where WPI is the WPI value for a particular location, X_i refers to component i of the WPI for that location, and w_i is the weight applied to that component. Each component is made up of a number of sub-components, and in the illustration presented in figure 5, all of the weights have been set at 1.

The computation of the WPI facilitates comparison between locations, and if repeated, can be used to assess progress (or otherwise), over time. By plotting the components together on a pentagram, the nature and relative importance of the water management issues at each site can be compared. Figure 5 provides an example of how the WPI varies between three different communities in northern Tanzania. In this example, we can see how this technique allows decision makers and communities to understand more clearly what particular components within the water sector need to be developed. For example, between these three communities, improvements in access are most needed in Samaria,[2] where the resource itself also needs to be developed. In the case of Kijenge, a peri-urban community, the capacity to manage water is relatively high, but benefits from the use of water need to be better developed. This can be contrasted with the situation in Nkoaranga, a rural community, where capacity to manage may be a little less, but the returns from water use are greater. This suggests that investment in capacity building would be particularly productive in Nkoaranga whereas development of more effective water use, and improved access, would be more applicable in each of the other locations studied. Displaying the information in this format makes it easier for policy makers and stakeholders to understand the full range of water-related issues, and how best to respond to these to

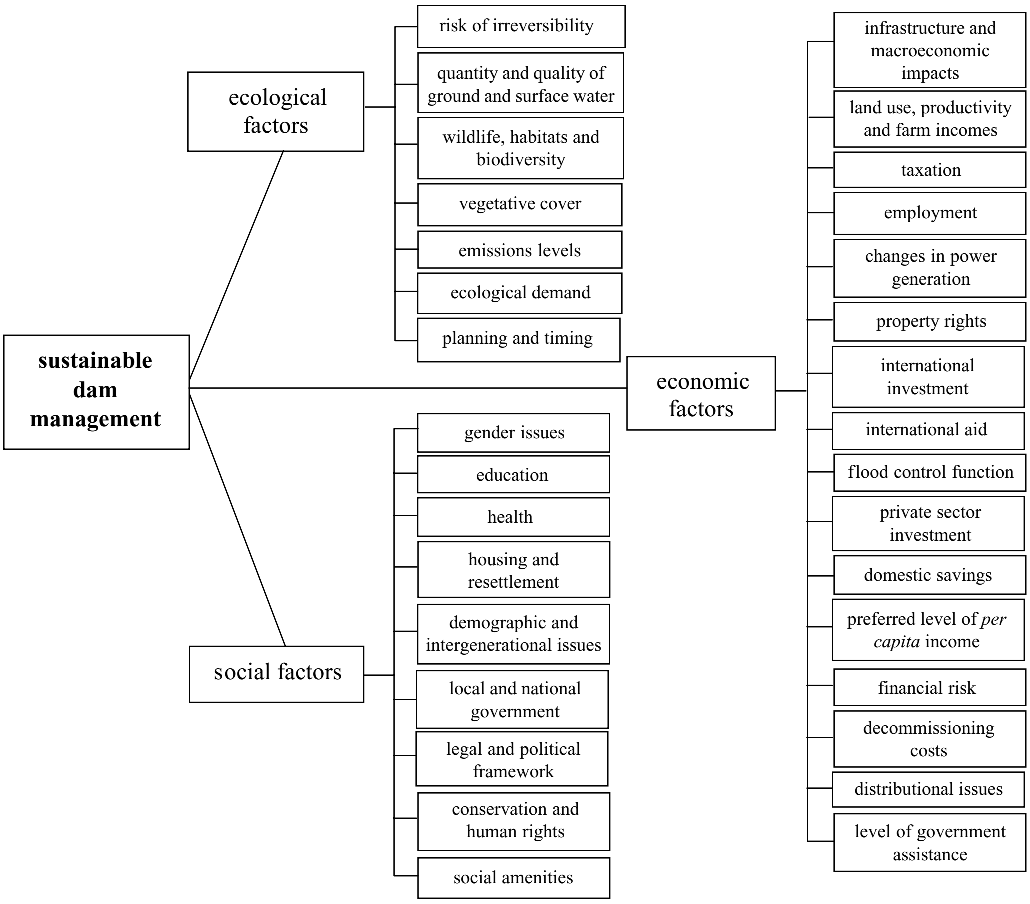

Figure 4. Using a multi-criteria approach for management of large dams.

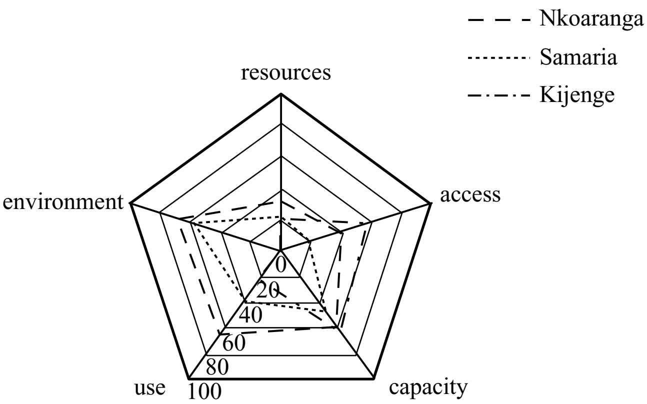

Figure 5. An illustration of the WPI applied to communities in Northern Tanzania (redrawn from Sullivan *et al.* 2002).

promote more effective and equitable development in the water sector.

The WPI is very useful for enabling the participation of various stakeholders in policy-making and valuation methodologies, a process that has been very widely advocated. Other approaches, such as 'deliberative democracy' (Jacobs 1997) in the area of environmental valuation, and 'multi criteria appraisal' (Stirling 1997) in the area of policy-making, have become more widely accepted in recent years. Deliberative democracy, like citizens' juries, presents the advantage of animating a debate and hence environmental awareness within a community, and this in turn constitutes a combination of environmental education and the construction of environmental values that are then communicable to policy makers.

3. ECOSYSTEM NEEDS

The idea that an allocation of water should be made for the natural environment was taken up at UNCED in 1982, where the governments of the United Nations made an ethical commitment to the environment in the form of the World Charter for Nature. This expresses absolute support of the governments for the principle of conserving biodiversity. It recognizes that every form of life is unique and warrants respect, regardless of its direct worth to humankind, and that the lasting benefits of nature depend on maintenance of essential ecological processes and life-support systems and upon the diversity of life forms (McNeely *et al.* 1990). The concept promotes conservation of ecosystems as a public good, independent of their utility as a resource and hence water rights to species and ecosystems. The declaration from the Second World Water Forum in The Hague 2000 highlighted the need to ensure the integrity of ecosystems through sustainable water resources management. The World Summit on Sustainable Development held in August 2002 in Johannesburg reinforces the role of environmental protection as a key pillar of sustainable development. South Africa has taken a lead in implementing the concept (Rowlston & Palmer 2002). Its Water Law states that 'the quantity, quality and reliability of water required to maintain the ecological functions on which humans depend shall be reserved so that the human use of water does not individually or cumulatively compromise the long-term sustainability of aquatic and associated ecosystems'. Tanzania is currently developing similar legislation that also gives high priority to ecosystem needs.

As the water resources of the world come under increasing pressure (see figure 1), their allocation between different uses becomes more critical. With a water crisis facing many countries, it seems an immense task just to manage water so that there is enough for people to drink, let alone for agricultural and industrial uses. Thus many people believe that providing water to other users, such as 'the environment', should be given low priority. Indeed, the situation is often presented as a conflict of competing demand, as though it were a matter of choice between water for people and water for wildlife. This ignores the benefits to mankind of functioning ecosystems, including natural resources (e.g. fish, timber and medicines), hydrological functions (e.g. flood protection and water quality improvement) and support of biodiversity. Water for ecosystems should thus be seen as water indirectly for people. Barbier *et al.* (1991) showed that the net economic benefits of water used to maintain resources and functions of the Hadejia-Nguru wetlands in Nigeria (agriculture, fishing, fuel wood) were many times the returns from using the water for intensive cereal irrigation.

Figure 6 summarizes the trade-off between allocating water to direct and indirect human uses. The upper part of figure 6 shows the impact of allocating water to natural aquatic ecosystems, which in turn provide valuable goods (e.g. fish), services (e.g. water regulation) and amenity–touristic–ethical value (landscape and species). In this case, the impact on the hydrological cycle is frequently positive, as, for example, when aquatic ecosystems improve water quality. Additionally, it satisfies the growing belief among many people that humans have a moral duty to protect wildlife through providing sufficient water to maintain flora and fauna.

The lower part of figure 6 shows the direct use of water through the development of highly managed systems, including reservoirs, intensive irrigation schemes, dams, river embankments and water purification plants. This has led to production of crops, industrial products, electricity, protection from floods and provision of clean water, thus improving economic and social security. However, this has often also caused negative impacts in the form of pollution. Clearly, the economic and social well-being of many people have been vastly improved. In addition, through the provision of food and water to starving and thirsty people in drought-stricken countries, technology has contributed to the ethical objectives of those who do not face this problem. Furthermore, there are many who suffer from development options, such as those who live off natural resources downstream. A major question is whether such highly managed systems are sustainable.

(a) *Optimizing the trade-off between direct and indirect use of water*

The important question is 'at what level to maintain the Earth's ecosystems?' The concept of sustainability suggests that we need to maintain the Earth's ecosystems so that they yield the greatest benefit to present generations, while maintaining the potential to meet the needs and aspirations of future generations. The problem is to decide how much water should be utilized directly for people for domestic use, agriculture and industry and how much water should be used indirectly by people to maintain aquatic ecosystems that provide environmental goods and services.

Figure 7 shows the problem conceptually as a trade-off between natural and highly managed systems. As natural systems are modified more and more, the benefits of the natural system decline; e.g. hydrological functions, products and biodiversity are lost. At the same time, benefits from the highly managed system increase; e.g. food production rises. It is suggested that the benefits from the highly managed systems reach a plateau, while the benefits of the natural system will decline to zero at some point. The total long-term benefits can be calculated by adding the benefits of the natural and highly managed systems. The total rises to a maximum before declining. It is at this point that the balance of the level of management is optimized. Obviously, the value that society places on goods and services and ethical considerations will determine the exact form of these curves. Indeed, the perceived benefits will vary between different groups and individuals. It is essential therefore that the costs and benefits to society of allocating water alternatively to maintain aquatic ecosystems and to support direct use in the form of agricultural, industrial and domestic uses are quantified.

(b) *Determining the water needs of aquatic ecosystems*

Two broad approaches have been adopted: objective-based water allocation and scenario-based allocation (Dunbar *et al.* 2003). As its name suggests, the objective-based approach requires pre-definition of objectives for an aquatic ecosystem, then the water required to achieve these is calculated. This may be driven, for example, by legal requirements such as the EU Directives that specify a target ecological status. The application of the objective-based approach by water managers forces the identification of threshold flows below which there would be significant changes in river ecology. For example, experts have suggested that in Australia the probability of having a healthy river falls from high to moderate when the hydrological regime is less than two-thirds natural (Jones 2002). In South Africa it is recognizes that different rivers will have different objectives. The Department of Water Affairs and Forestry classifies rivers according to four target classes, A–D (table 2). The environmental flow allocated to the river depends on the target class and is defined by the BBM (Tharme & King 1998). The basic premise is that riverine species are reliant on basic elements (building blocks) of the flow regime, including low flows and floods (that maintain the sediment dynamics and geomorphological structure of the river). An acceptable flow regime can thus be constructed by combining these building blocks. This approach makes strong use of an expert group of physical and biological scientists who, by following a series of structured stages, assess available data and model outputs and use their combined professional experience to come to a consensus on the building blocks of the flow regime. The BBM is currently routinely used in South Africa to comply with the 1998

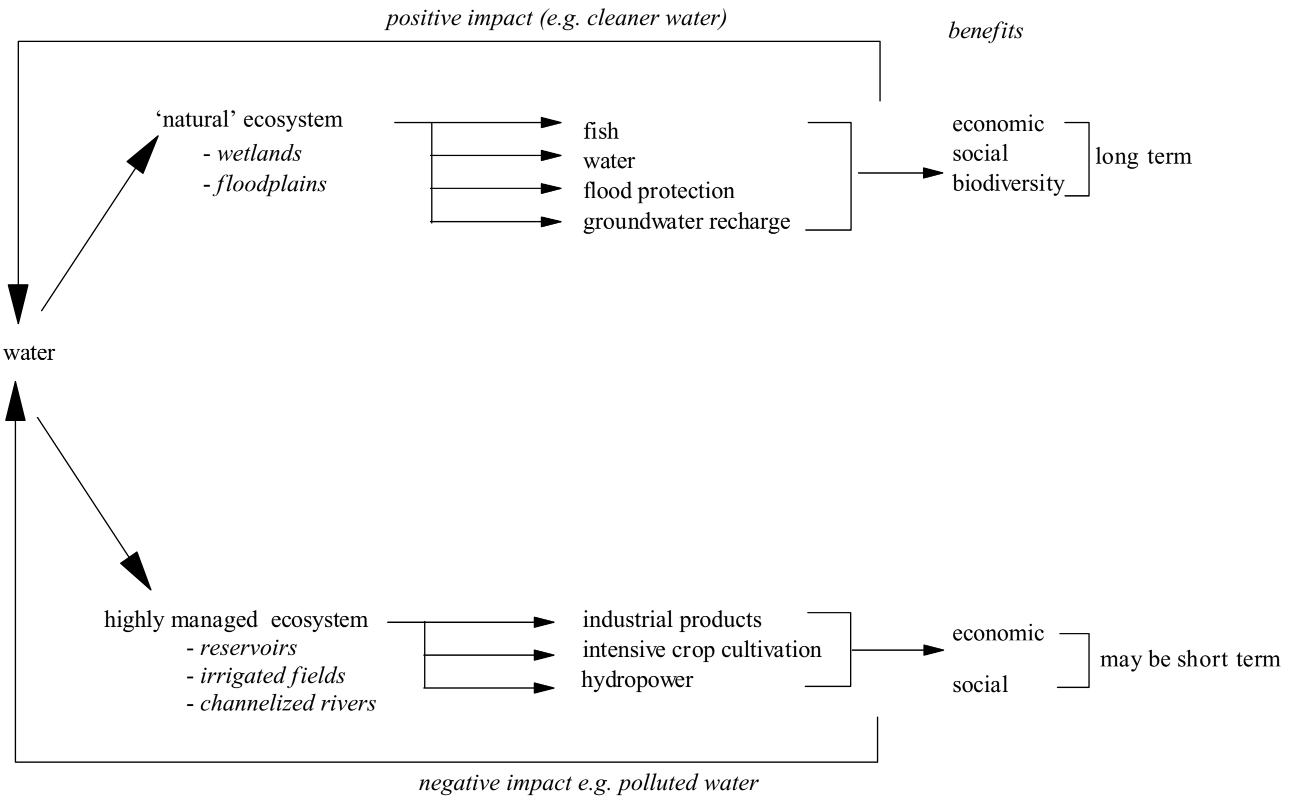

Figure 6. Natural and non-natural ecosystem benefits.

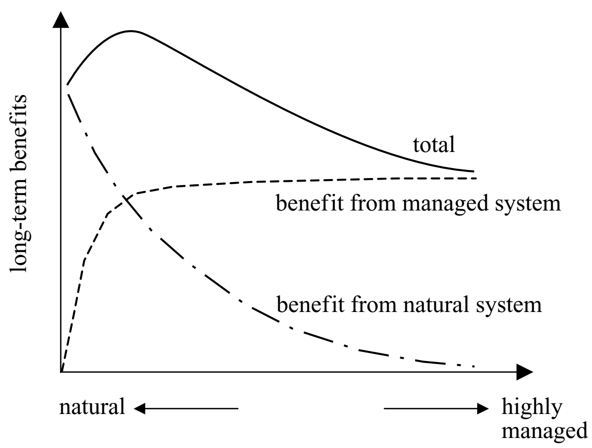

Figure 7. Maximizing benefits from freshwater ecosystems.

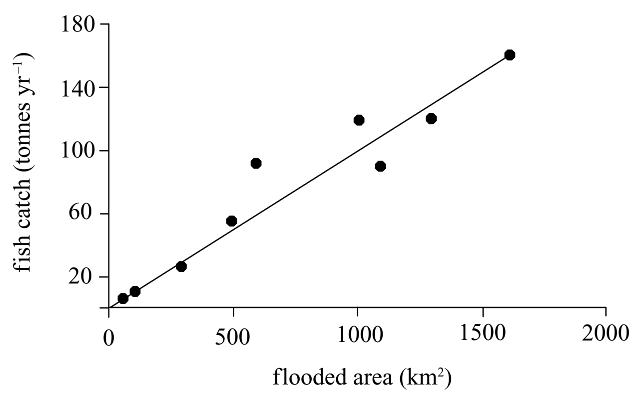

Figure 8. Relationship between flooded area and fish catch (redrawn from Welcomme 1996).

Water Act, has been applied in Australia, and is also being tried in the USA.

The major problem with the objective-based approach has been the difficulty in identifying critical threshold flow levels. Many relationships between river flow and ecological indices are straight lines or smooth curves, such as the change in fish abundance with flow (Sheldon *et al.* 2000). These do not provide evidence of specific flow thresholds to support ecological objectives. Welcomme (1996) found a linear relationship between flooded area and fish catch (figure 8) on African floodplains, thus there is no clear threshold point below which the flooded area is insufficient to maintain the fish population. Some studies have indicated a continuum of change in ecological communities, such as macro-invertebrate, with alteration of flow regime (Extence *et al.* 1999). In the face of uncertainty, adaptive management approaches have been adopted in many countries including the USA and South Africa, where initial river flow thresholds are set and then adjusted as the aquatic ecosystem responds.

Where no specific ecological objectives have been set, which is the case for most aquatic ecosystems, scenario-based environmental flow allocation provides an alternative approach. In this approach, various river flow scenarios are used as the basis of negotiation between stakeholders. Scenarios often include a natural flow regime and several flow regimes produced by different degrees of water abstraction or impoundment of the river. For example, to assist the Lesotho government in setting environmental flows from dams within the Lesotho Highlands Development Project, Brown & King (2000) developed the DRIFT approach. Like the BBM and other similar methods, it is an holistic approach that addresses

Table 2. Ecological management classes (DWAF 1999).

class	description
A	negligible modification from natural conditions, negligible risk to sensitive species.
B	slight modification from natural conditions, slight risk to intolerant biota.
C	moderate modification from natural conditions, especially intolerant biota may be reduced in number and extent.
D	high degree of modification from natural conditions, intolerant biota unlikely to be present.

all aspects of the river ecosystem. DRIFT produces, for various scenarios, the economic benefits of selling impounded water to South Africa and the impacts on dependent communities of the resulting degraded river ecosystems. In the UK, Dunbar *et al.* (2000) modelled the impact on salmonid fish habitat of five different groundwater abstraction options in the River Wylye to assist in setting abstraction licences. This approach is designed to address trade-offs between the needs of the abstractor and the environment. The advantage is that the scientists do not need to define hydro-ecological thresholds and various stakeholders can be involved in the process. The disadvantage is that the approach is inconsistent, as different groups of stakeholders reach different decisions.

The water required to meet specific ecosystem objectives, such as conservation of particular species or communities is a complex issue that has been the subject of considerable study worldwide. Much of the focus has been on minimum flow requirements. However, there is increasing recognition that the magnitude, duration, frequency and timing of many elements of the flow regime of rivers are important to their ecology rather than just low flows (Poff *et al.* 1997; Richter *et al.* 1997). For example, Booker *et al.* (2003) found that in urban rivers in Birmingham, UK, fish habitat was critically low at high flows since velocities were higher than maximum fish swimming speeds because refuges had been removed through channelization aimed at reducing flooding.

Some species have specific requirements during a particular life stage. For example, Acacia trees in the riverine forests of the Indus river valley require inundation from flood water for their moisture, which also brings important nutrients. At least in their early stages of growth, the trees must be flooded for at least 10 days per year. Once acacias are *ca*. 8–10 years old their roots are normally able to reach the permanent water table. Even tree species that grow in saline water, such as mangroves, have requirements for freshwater. For example, ecological studies of the Indus delta by the Sindh Forestry Department (Qureshi 2001) estimated that each 100 acres (40 ha) of mangrove forest requires 1 cusec (0.028 $m^3 s^{-1}$) of water during July and August to remain healthy and support the associated fisheries. For the estimated 260 000 hectares of mangroves a total volume of 27 MAF (33 300 million m^3) would be needed. The floodplain forests upstream of the delta have a different water requirement. They need to be inundated at least twice in 5 years to enable saplings to become established. By combining these requirements, an ecological minimum flow regime can be defined, in terms of volume, distribution through the year and inter-annual variability, to maintain the ecosystem of the lower Indus.

In some cases, biological response models are employed to predict changes in physical habitat for particular species or life stages that result from alterations to the flow regime. The most widely used method is the PHABSIM initiated by the US Fish and Wildlife (Bovee 1986) and developed for use in other countries including UK, New Zealand, France and Norway (Parasiewicz & Dunbar 2001). PHABSIM assumes that a given species has preferences for certain habitat characteristics, such as water depth or flow velocity. A hydraulic model within PHABSIM, which requires calibration using field measurements, determines the spatial variation in depths and velocity and predicts how this changes with flow. A key output from PHABSIM is a quantitative relationship between in-stream physical habitat and river flow for key species (Elliott *et al.* 1996). PHABSIM has been used to estimate the ecological effects (in terms of available physical habitat) for historical or future anticipated changes in flow caused by abstraction, dam construction or climate change. However, it does not normally consider indirect impacts, for example reduced river flows may increase concentrations of pollutants or reduce dissolved oxygen.

4. INTEGRATED WATER RESOURCES MANAGEMENT

Many of the problems of environmental degradation resulting from human use of water resources have come about owing to a lack of understanding of how changes in the quantity and quality of water at one point in space and time can affect aquatic and their associated ecosystems at another point in space and time. In recognition of this, Falkenmark *et al.* (1999) has written about upstream and downstream impacts of water use and the need to understand better the linkages between water and the landscape that it flows through. Falkenmark has also highlighted three key past omissions as being:

(i) the focus on river ('blue') water that locks the debate into irrigated food production and ignores the important contribution of rain-fed ('green') production;
(ii) ignoring what happens to water after its use—i.e. alterations to the downstream quantity and quality of water and its potential reuse; and
(iii) the need to consider environmental–ecosystem water requirements.

Many authors (Rijsberman & Molden 2001; Wallace & Gregory 2002) have also pointed out that it is necessary to not only understand the linkages within the physical water system and the ecosystems it supports, but also the linkages with the social systems that depend on and manage the water resources. This mixture of physical and social issues is very complex and it is therefore vital to have some frame of reference within which this complexity can be addressed. Within the hydrological community, this frame of reference is the catchment, a generally well-defined geographical unit where it is possible to describe water inputs and outputs as well as flows within the system. Using a catchment as the unit of study allows the impacts of change in one part of the catchment to be predicted in another part of the catchment. For example, deforestation or afforestation of the uplands in a catchment can alter flows and water quality downstream, an issue that has long been studied by hydrologists (Calder & Newson 1979). Catchment frameworks have also been used in the design and impact assessment of dams on rivers (Acreman *et al.* 2000). The approach has been so useful that it has led to the concept of IWRM, i.e. using a catchment framework to assess and manage the complete array of water supply and demands within the specified basin. This approach is considered as vital, especially in water-scarce areas where the competition for limited water resources is most acute. These areas tend to be ones where agricultural productivity is low and largely rain-fed. However, Rijsberman & Molden (2001) have pointed out that water used by rain-fed agriculture can also impact downstream aquatic ecosystems if the water evaporated by the crops reduces run-off. Rijsberman & Molden (2001) also advocate the co-management of both rain-fed and irrigated agriculture within a catchment context. Using this approach they conclude that it would be possible to meet future food production requirements without increasing the allocation of water to agriculture, but that this would require productivity improvements in both rain-fed and irrigated agriculture that exceed (by a factor of 2) current performance.

It is encouraging to see the concept of IWRM, which has existed for some time in scientific circles, being recognized and used by water managers and policy makers. For example, water managers in the UK are now using catchments as the operational water management unit. Furthermore, new water legislation in both South Africa and China has adopted the catchment as the basis for their new water laws, requiring new water administration organizations to operate on a catchment basis. The new water law in South Africa also introduces the concept of legally protected environmental flows, effectively giving 'water rights' to the environment. As the use of catchments for IWRM is a major step forward, we need to consider their use in contributing to the issue of the sharing of water between society and nature. How can we develop catchment-based IWRM to help address this issue?

The physical linkages within a catchment offer the possibility of quantifying the interactions between different water users (e.g. the impacts of water abstraction for irrigation on river flows downstream). To do this clearly requires some hydrological knowledge of the catchment, which in most parts of the world is either incomplete or entirely missing. However, if we wish to predict the full range of environmental impacts, we also need to understand the relationships between river flow regimes and the in-stream aquatic and riparian ecosystems that they support. This is a new and very challenging area of physical science, where there are some early examples of progress (e.g. the PHABSIM work referred to earlier). The key challenge is to quantify the impact of sub-optimal water supply on aquatic ecosystems. This evaluation must take into account the temporal and spatial variations of aquatic ecosystem requirements, not least because this information may be vital in finding the best match between ecosystem and human requirements. Only then can rational decisions be made on how best to apportion limited water supplies between different parts of the aquatic ecosystem and society.

The critical timing of water requirements for ecosystems and humans has been illustrated in the Diawling National Park, in the delta of the River Senegal in Mauritania. This area is affected by seasonal variations in water availability and salinity, which has generated particular vegetation types, such as mangroves with associated species, including penaeid shrimp and mullet (Hamerlynck 2001). In the late 1980s, the delta was separated hydrologically from the river by construction of the Diama dam and right-bank embankment. These maintain water levels in the Senegal River for gravity irrigation and navigation, but caused degradation of biodiversity of the Park and loss of natural resources, including grazing and fisheries, in the buffer zone on which local communities depend. Thus, construction of the embankments led to increased poverty of these people. Restoration of the Diawling Park involved releasing water through the embankment via a sluice gate. On 1 July initial releases are made to dampen the soil, simulating rainfall. On 1 August releases are increased so that the water level rises at a maximum of 1 cm d^{-1}, so that the growth of grasses, such as *Sporobolus robustus* and *Echinocloa colona* can keep pace. The grass provides habitat for fish that spawn on the floodplain and for the nesting of crowned cranes (*Balearica regulorum*). Annual fish production increases with flooded area by *ca.* 100 kg ha^{-1}, providing food and income for many local people. The flooded wetlands also provide habitat for many thousands of migratory birds. After 45 days of inundation, salt has been leached from the soils and the water is allowed to drain off, to prevent colonization by unwanted species such as *typha* and cypergrasses. In the dry season the *Sporobolus* is exploited for the production of mats, providing the main source of income for local women. The *Echinocloa* provides excellent grazing for the thousands of cattle visiting the delta. This is a good example of integrated water resources management.

Where surface water dominates, the catchment provides the most appropriate management unit. However, frequently the underlying aquifer does not coincide with the surface river basin. Thus, where groundwater plays a significant role, a group of basins overlying the aquifer may constitute the appropriate unit of water resource management. For issues where air quality is influential, such as acid rain, the 'airshed' (as apposed to the watershed) will be more appropriate implying an area encompassing the source areas, which may be industries in the UK and affected areas in Scandinavia (Acreman 1998).

In the broader context of the sharing of water between society and nature, the biggest challenge is to see how the catchment framework can be used for the non-physical (social, legal, economic and institutional, etc.) factors that are important to water use and its management. The institutional dimension is important since water management organizations need to be able to link individual waters users and sub-catchment systems (e.g. dams, irrigation schemes, towns, etc.) to the complete catchment level. Economic factors cross catchment boundaries and it is important to be able to deal with this. For example, a water-scarce area with sufficient money may decide to import some (or all) of its basic food requirement, thereby negating the need to use catchment water supplies to grow food. This has been referred to as importing 'virtual water' (Allan 1998*a*,*b*). The key point is that an interaction between internal catchment water resources and external economics needs to be understood and introduced into the catchment management.

Legal issues can arguably be the most dominant factor in determining how water is managed. For example, who owns water and what price should be paid for it are major issues that are usually determined through legislation. However, how much hydrological and ecological information is used in setting and enforcing water laws? Furthermore, how well is legislation matched to catchment boundaries and how do you deal with catchments that cross legal boundaries (e.g. the Danube in Europe and the Mekong in Asia, etc.)? The interface between water and law is clearly another major challenge that needs to be met for the successful implementation of good water resources management. Finally, there are many complex social aspects to water. These range from cultural and religious beliefs that affect the use of water to modern societies' expectations of water resources for recreation and aesthetic value.

Societal priorities for water need to recognize that huge numbers of people throughout the world are today still suffering the burden of poverty. Any new approach to water management must explicitly address this problem, since a typical characteristic of poverty is lack of access to adequate water supplies. Since the proportions needed for domestic requirements are relatively small, accounting only for some 3–4% of total *per capita* requirements (Falkenmark 1997), this suggests that losses resulting from small adjustments in allocations to other uses will have a more than proportionate impact on domestic water supply. Bringing about more effective redistribution of water to reduce poverty is therefore very possible. By demonstrating the small degree of change needed to achieve real impacts on poverty alleviation, policy makers may be persuaded to adopt both the appropriate political will and the necessary finance needed to bring about real change.

Taking the water management problem as a whole, however, allocations between sectors need to be examined, with the potential goal of highlighting current inefficiencies and inequities in sectoral water use. By highlighting such inefficiencies, attention can be drawn to weaknesses in existing systems, and the opportunities to improve water management through its more equitable reallocation. There is little doubt that water allocations for effective aquatic ecosystem health are important to the maintenance of the ecological integrity of human life-support systems. As a result, we need to work out how best to share water between people and nature. While it is vital that adequate water be reserved for the maintenance of ecological services, the quantities needed for this are not yet fully understood. However, it is likely that ecological requirements will represent a significant proportion of the available water resource, although this will mostly be non-consumptive use. To incorporate this ecological allocation into water management, new approaches are needed to quantify the effects of non-optimal supply on aquatic ecosystems. When these are better known, along with human requirements and the efficiencies of the different sectors, a more equitable approach to the sharing of water between society and nature should be possible.

5. CONCLUDING REMARKS

Despite the abundance of water on the Earth, it is becoming clear that the relatively small proportion that is fresh and accessible is coming under increasing pressure as the world population rises. Human requirements for water have greatly increased in the past 50 years and look set to continue to increase for at least another 50 years. These requirements are dominated by the water used in irrigated agriculture, where significant productivity gains (yield per unit of water used) are required to limit the water allocation to this sector to acceptable levels. Conversely, basic human requirements for drinking and other domestic purposes is a very small proportion of the total water used and hence it should be possible to meet these vital needs without much impact on other sectors.

As human water requirements grow, the water left to support aquatic ecosystems is reduced and the already considerable ecosystem impacts will continue to rise. However, it is now widely recognized that supplying sufficient water to aquatic ecosystems is essential to maintaining the very essence of life on Earth. Ecosystems provide direct support of humans through natural resources and hydrological functions and they are a fundamental part of the social structure of many rural communities. In line with the principles of sustainable development, there is also a growing acceptance that we have a moral and ethical duty to maintain our fellow species on the planet. Science can provide an insight into the fundamental principles of ecosystem functioning and its interactions with water. We are beginning to gain a better understanding of the water requirements of species and communities and about the thresholds that define critical levels of water supply to maintain ecosystem health. Nevertheless, in many parts of the world aquatic ecosystems have become highly managed and it is not possible, or even in many cases desirable, to return them to a natural state and only small areas will ever be conserved as true wilderness. The desired state of our ecosystems will inevitably be guided primarily by social choice. Scientific principles can then be employed to define the necessary water requirements. Whether these requirements can be met will depend upon a complex interaction between law, policies, economics and politics.

Equity has become an important concept in water allocation in the past decade. As pressure on water resources increases, ensuring equity between economic, social and environmental sustainability will become a major challenge. Over the past 100 years, economic development has been dominated by infrastructure, with water allocation focused on intensive agriculture, hydropower generation and industrial and domestic supply. There has been a repeated tendency to neglect the needs of the rural poor, who, more than any others, are dependent on natural resources and functions of ecosystems. More equitable allocations of water for poor peoples' needs have to be met by redistribution from other sectors, although the quantities involved are relatively small. At the same time, conservation of ecosystems and rare species has often had the lowest priority. It has now become clear that the long-term survival of human and biological diversity on Earth will be dependent on a new paradigm of equitable allocation between our economic, social and ecological needs. More equitable sharing of water resources between society and nature will require values to be placed on both human and aquatic ecosystem requirements. These values will need to be incorporated into macro-economic policies so that more rational decision-making about water allocations can be made, not only to meet the needs of our current population but also to meet the needs of the continually growing future generations.

Catchment-based IWRM provides a framework within which water resources can be managed more holistically. This makes it possible for human and aquatic ecosystem water needs to be better understood, highlighting areas where water use efficiency can be improved or where limited water is best used. More effective natural resource accounting in general, and water accounting in particular, would provide a solid foundation on which more sustainable and equitable policies can be built. The development of these types of accounts have been resisted by many large-scale enterprises, as they anticipate the associated cost implications. However, through a process of continued institutional strengthening, it will become more likely that better representation of different groups will be achieved in decision making, enabling a wider range of views to be considered. There is little doubt that such institutional development is crucial to the water sector, and by contributing to this process, catchment-based co-management can ensure that more effective ways of representing the interests of both people and nature can be achieved.

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ENDNOTES

[1]This is based on a 30% reduction overall of water required for food production, following a switch to a lower meat diet.

[2]In this community, it was found that households spend, on average, between 6 and 7 h d^{-1} collecting water for their domestic needs (Sullivan *et al.* 2002).

REFERENCES

Abramovitz, J. N. 1997 Valuing Nature's Services. In *State of the World—progress towards a sustainable society*, pp. 92–114. Washington, DC: Worldwatch Institute.

Acreman, M. C. 1998 Principles of water management for people and the environment. In *Water and population dynamics* (ed. A. de Shirbinin & V. Dompka), pp. 25–48. Washington, DC: American Association for the Advancement of Science.

Acreman, M. C. 2001 Ethical aspects of water and ecosystems. *Water Policy J.* **3**, 257–265.

Acreman, M. (and 10 others) 2000 *Managed flood releases: issues and guidance*. Wallingford, UK: Institute of Hydrology, for the World Commission on Dams.

Alcamo, J., Döll, P., Kaspar, F. & Siebert, S. 1997 *Global change and global scenarios of water use and availability: an application of WaterGAP1.0*. Kassel, Germany: Centre for Environmental Systems Research, University of Kassel.

Alcamo, J., Henrichs, T. & Rosch, T. 2000 *World water in 2025—global modelling scenarios for the World Commission on water for the 21st Century*. Kassel World Water Series, Report No. 2. Kassel, Germany: Center for Environmental Systems Research, University of Kassel.

Allan, J. A. 1998*a* Moving water to satisfy uneven global needs: 'trading' water as an alternative to engineering it. *ICID* J **47**, 1–8.

Allan, J. A. 1998*b* Virtual water: a strategic resource: global solutions to regional deficits (Editorial). *Ground Water* **36**, 546–546.

Barbier, E. B., Adams, W. M. & Kimmage, K. 1991 *Economic valuation of wetland benefits: the Hadejia-Jama'are Floodplain, Nigeria*. London Environmental Economics Centre Paper DP 91-02. London: International Institute for Environment and Development.

Berkes, F. & Folke, C. 1994 Investing in natural capital for sustainable use. In *Investing in natural capital: the ecological economics approach to sustainability* (ed. A. Jansson, M. Hammer, C. Folke & R. Costanza). Washington, DC: Island Press.

Biswas, A. K. 1989 Sustainable water development and management. A synthesis. *United Nations Environ. Programme* **35**, 1639–1655.

Booker, D. J., Dunbar, M. J., Shamseldin, A., Durr, C. S. & Acreman, M.C. 2003 Physical habitat assessment in urban rivers under future flow scenarios. *J. Chartered Inst. Water Environ. Mngmt.* (In the press.)

Bovee, K. D. 1986 Development and evaluation of habitat suitability criteria for use in the instream flow incremental methodology. Instream Flow Information Paper No. 21. *US Fish Wildl. Serv. Biol. Rep.* **86**, 1–235.

Brown, C. & King, J. 2000 Environmental flow assessment for rivers: a summary of the DRIFT process. Southern Waters Information Report 01/00. South Africa: Mowbray.

Calder, I. R. & Newson, M. D. 1979 Land-use and upland water resources in Britain—a strategic look. *Water Res. Bull.* **15**, 1628–1639.

Chambers, R. 1995 *Poverty and livelihoods: whose reality counts? IDS Discussion Paper 347*. Brighton: IDS.

Chiuta, T. M. 1995 Indigenous knowledge systems for wetland conservation: Barotse Floodplain, Southern Africa. *IUCN Wetlands Programme Newslett.* **11**, 6–7.

Costanza, R. (and 15 others) 1997*a* The value of the world's ecosystem services and natural capital. *Nature* **387**, 253–260.

Costanza, R., Cumberland, J., Daly, H., Goodland, R. & Norgaar, R. 1997*b An introduction to ecological economics*. Boca Raton, FL: CRC Press.

Daly, H. 1999 *Ecological economics and the ecology of economics*. Cheltenham, UK: Edward Elgar.

Desai, M. 1995 *Poverty, famine and economic development*. Aldershot, UK: Edward Elgar.

DFID (Department for International Development) 2002 *Poverty and environment*. London: DFID.

Dunbar, M. J., Gowing, I. M., Linstead, C. & Maddock, I. 2000 PHABSIM Investigations on the River Wylye: PHABSIM model calibration and time series analysis. Environment Agency Contract SWCON61. Unpublished contract report to the Environment Agency. Exeter, UK: Environment Agency.

Dunbar, M. J., Acreman, M. C. & Kirk, S. 2003 Environmental flow setting in England and Wales: strategies for managing abstraction in catchments. *J. Chartered Inst. Water Environ. Mngmt.* (In the press.)

DWAF (Department of Water Affairs and Forestry) 1999 *Resource directed measures for protection of water resources*. Pretoria, South Africa: DWAF.

Elliott, C. R. N., Johnson, I. W., Sekulin, A. E., Dunbar, M. J. & Acreman, M. C. 1996 *Guide to the use of the Physical Habitat Simulation System*. Report to the National Rivers Authority. Wallingford, UK: Institute of Hydrology.

Extence, C., Balbi, D. M. & Chadd, R. P. 1999 River flow indexing using British benthic macro-invertebrates: a framework for setting hydro-ecological objectives. *Regulated Rivers Res. Mngmt* **15**, 543–574.

Falkenmark, M. 1997 Meeting water requirements of an expanding world population. *Phil. Trans. R. Soc. Lond.* B **352**, 929–936. (DOI 10.1098/rstb.1997.0072.)

Falkenmark, M. 2002 *No freshwater security without major shift in thinking*. Stockholm: Stockholm International Water Institute.

Falkenmark, M. (and 11 others) 1999 *Water: a reflection of land use*. Stockholm, Sweden: Swedish Natural Science Research Council.

Faucheux, S. & O'Connor, M. 1998 *Valuation for sustainable development*. Cheltenham, UK: Edward Elgar.

Fischer, G. & Heilig, G. K. 1997 Population momentum and the demand on land and water resources. *Phil. Trans. R. Soc. Lond.* B **352**, 869–889. (DOI 10.1098/rstb.1997.0067.)

Gleick, P. 2000 *The World's water 2000–2001*. London: Island Press.

Hamerlynck, O. 2001 Flooding regime of the Diawling Park Mauritania. Summarised in Acreman, M. (and 10 others) 2000 *Managed flood releases: issues and guidance.* Wallingford, UK: Institute of Hydrology, for the World Commission on Dams.

Hewlett, J. D. & Hibbert, A. R. 1967 Factors affecting the response of small watersheds to precipitation in humid regions. In *Forest hydrology* (ed. W. E. Sopper & H. W. Lull), pp. 275–290. Oxford: Pergamon.

Jacobs, M. 1997 Deliberative democracy. In *Valuing nature. Ethics, economics and the environment* (ed. J. Foster). London: Routledge.

Jones, G. 2002 Setting environmental flows to sustain a healthy working river. *Watershed*, February 2002. Canberra, ACT: Cooperative Research Centre for Freshwater Ecology. See http://freshwater.canberra.edu.au.

McCully, P. 1996 *Silenced rivers: the ecology and politics of large dams*. London: Zed Books.

McNeely, J. A., Miller, K. R., Reid, W. V., Mittermeier, R. A. & Werner, T. B. 1990 *Conserving the world's biodiversity*. Gland, Switzerland: IUCN.

Maidment, D. R. 1992 *Handbook of hydrology*. New York: McGraw-Hill.

Noel, J. F. & O'Connor, M. 1998 Strong sustainability and critical natural capital. In *Valuation for sustainable development: methods and policy indicators* (ed. S. Faucheux & M. O'Connor). Cheltenham, UK: Edward Elgar.

Norgaard, R. B. 1991 Sustainability as intergenerational equity. Asia Regional Series, Report IDP-97. Washington, DC: World Bank.

Opschoor, J. B. 1998 The value of ecosystem services: whose values? *Ecol. Econ.* **25**, 41–43.

Parasiewicz, P. & Dunbar, M. J. 2001 Physical habitat modelling for fish: a developing approach. Large rivers 12 2-4. *Arch. Hydrobiol.* **135**(Suppl.), 239–268.

Penning de Vries, F. W. T., Rabbinge, R. & Groot, J. R. R. 1997 Potential and attainable food production and food security in different regions. *Phil. Trans. R. Soc. Lond.* B **352**, 917–928. (DOI 10.1098/rstb.1997.0071.)

Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegaard, K. L., Richter, B. D., Sparks, R. E. & Stromberg, J. C. 1997 The natural flow regime. *Bioscience* **47**, 769–784.

Powell, J. M. 1998 Interregional environmental policy in Australia's Murray Darling Basin. In *The arid frontier* (ed. H. Bruins & J. Lithwick). Dordrecht, The Netherlands: Kluwer.

Qureshi, T. 2001 Indus delta flow needs. Summarised in Acreman, M. (and 10 others) 2000 *Managed flood releases: issues and guidance*. Wallingford, UK: Institute of Hydrology, for the World Commission on Dams.

Richter, B. D., Baumgartner, J. V., Wigington, R. & Braun, D. P. 1997 How much water does a river need? *Freshwater Biol.* **37**, 231–249.

Rijsberman, F. R. & Molden, D. 2001 Balancing water uses: water for food and water for nature. Draft thematic background paper. In *Int. Conf. on Freshwater, Bonn 2001*. Colombo, Sri Lanka: IWMI.

Roodman, D. 1999 *The natural wealth of nations. Harnessing the market and the environment*. London: Earthscan.

Rowlston, W. S. & Palmer, C. G. 2002 Processes in the development of resource protection provisions on South African Water Law. In *Proc. of the Int. Conf. on Environmental Flows for River Systems, Cape Town, 2002*. Rondebosch, South Africa: University of Cape Town.

Rubin, D. & Topping, D. 2002 *EOS* 83, 273.

Sheldon, F., Thoms, M. C., Berry, O. & Puckridge, J. 2000 Using disaster to prevent catastrophe: referencing the impacts of flow changes in large dryland rivers. Regulated rivers. *Res. Mngmt* **16**, 403–420.

Shiklomanov, I. A. 1991 The world's water resources. In *Proceedings of the International Symposium to Commemorate 25 Years of the IHP*, UNESCO/IHP, pp. 93–126. Paris: UNESCO.

Smakhtin, V., Revenga, C. & Doll, P. 2003 *Environmental water requirements and global water availability*. Sri Lanka: International Water Management Institute. (In preparation.)

Smith, A. 1786 *An enquiry into the nature and causes of the wealth of nations*, 4th edn. London: A. Strahan & T. Cadell.

Stiglitz, G. 1979 A neoclassical analysis of the economics of natural resources. In *Scarcity and growth reconsidered* (ed. V. K. Smith). Baltimore, MD: Johns Hopkins University Press.

Stirling, A. 1997 Multi criteria mapping. Mitigating the problem of environmental valuation. In *Valuing nature. Ethics, economics and the environment* (ed. J. Foster). London: Routledge.

Sullivan, C. A. 2002 Using an income accounting framework to value non-timber forest products. In *Valuation methodologies* (ed. D. Pearce). Cheltenham, UK: Edward Elgar.

Sullivan, C. A., Meigh J. R. & Fediw, T. 2002 Derivation and testing of the water poverty index, phase 1, final report. Wallingford, UK: Centre for Ecology and Hydrology.

Tharme, R. E. & King, J. M. 1998 Development of the building block methodology for instream flow assessments and supporting research on the effects of different magnitude flows on riverine ecosystems. Report to the Water Research Commission (no. 576/1/98). Rondebosch, South Africa: Freshwater Research Unit, Zoology Department, University of Cape Town.

UNDP 2001 Human development report 2001. New York: Oxford University Press.

Vorosmarty, C. J., Green, P., Salisbury, J. & Lammers, R. B. 2000 Global water resources: vulnerability from climate change and population growth. *Science* **289**, 284–288.

Wallace, J. S. 2000 Increasing agricultural water use efficiency to meet future food production. *Agric. Ecosystems Environ.* **82**, 105–119.

Wallace, J. S. & Gregory, P. J. 2002 Water resources and their use in food production systems. *Aquatic Sci.* **64**, 1–13.

Welcomme, L. 1996 Some general and theoretical considerations on the fish yield of African rivers. *J. Fisheries Biol.* **8**, 351–364.

World Resources Institute 2000 World Resources 2000–2001. People and ecosystems: the fraying web of life (in collaboration with the UNDP, UNEP and the World Bank). Washington, DC: World Resources Institute.

GLOSSARY

BBM: building block methodology
DRIFT: downstream response to imposed flow transformation
EU: European Union
GDP: gross domestic product
HDI: human development index
IWRM: integrated water resources management
PHABSIM: physical habitat simulation
WPI: water poverty index