Chapter 7

Freshwater Ecosystem Services

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BOXES

FIGURES

TABLES

Main Messages

Fresh water can make a greater contribution to human well-being if society improves the design and management of water resource infrastructure, establishes more inclusive governance and integrated approaches to water management, and adopts water conservation technologies, demand management, and market-based approaches to reallocation that increase water productivity. Rising human population and levels of socioeconomic development have led to a rapid rate of water resource development and the replacement of naturally occurring and functioning systems with highly modified and human-engineered systems. Meeting human needs for freshwater provisioning services of irrigation, domestic water, power, and transport has come at the expense of inland water ecosystems—rivers, lakes, and wetlands—that contribute to human well-being through recreation, scenic values, maintenance of fisheries and biodiversity, and ecosystem function.

The principal challenge is to balance these competing demands by acquiring the necessary institutional and financial resources and applying existing technologies, processes, and tools in order to increase the overall productivity of water for society. Agreement on rights and responsibilities with respect to the allocation and management of freshwater services is essential to reconcile diverging views on the degree of public and private participation. As agriculture comprises the major use of water resources globally, the choices between market reallocation and public/private investments in conservation will largely determine whether timely and cost-effective solutions will be found. The potential for climate change to alter availability and distribution of water supplies could be a further complicating factor.

In order to balance competing demands, it is critical that society explicitly agrees on ecosystem water requirements (environmental flows). Determination of ecosystem water requirements involves a societal decision of the desired condition of an ecosystem informed by data on the relationship between hydrology and ecosystem services, and followed by a cognitive or technical response to determine the water quantity and quality necessary to meet articulated objectives. This process is likely to be most successful when it is a collaborative one involving scientists, natural resource managers, and other stakeholders influenced by changes in the availability of the services provided by an ecosystem. Success in achieving outcomes is likely to take time to occur and measure. Any decision on ecosystem water requirements needs to be supported in national and regional water management policies and implemented through an adaptive management approach.

The shift from development of new supplies to emphasis on the reallocation of existing supplies, and integrated water resources management is fundamentally an issue of governance, which provides an entry point for broader policy reforms, as it implies the need to evaluate trade-offs among multiple and often conflicting uses across sectors, within the context of the entire flow regime. Key challenges in governance of freshwater services include democratic decentralization of decision-making processes and recognition or establishment of appropriate forms of property rights and responsibilities, which enables stakeholders to be adequately represented and to hold decision-makers accountable, which can make a difference in whether or not objectives are achieved. Appropriate forms of governance are also necessary to ensure equity, which is a fundamental enabling condition in the use of regulatory and/or market-based incentives for protecting freshwater services. River basin organizations can play an important role in facilitating cooperation and reducing transaction costs of large-scale responses.

Given the diversity of conditions in river basins, the more effective kinds of arrangements may be those that have evolved in response to site-specific conditions and extreme events, as these raise awareness of impacts and provide an opportunity to open policy debates. A key constraint will be negotiating with those who have vested interests in existing arrangements and who are resistant to change. Such reforms may be difficult and have high transaction costs, but may gain momentum from broader-scale social, economic, and political changes from which they may be inseparable, such as occurred after the fall of apartheid South Africa. They may also have added benefits—of strengthening democratic institutions and contributing to political stability, as occurred in the Danube Basin following the cold war period. Quality of information, obtained through independent and transparent assessment, is critical for increasing stakeholder confidence and, therefore, willingness to pay or otherwise cooperate in responses, and for identifying specific barriers to implementation.

Economic incentives have the potential to unlock significant supply- and demand-side efficiencies while providing cost-effective reallocation between old (largely irrigation) and new (largely municipal and instream) uses. Historic allocations of water have rarely taken account of its scarcity value or of the economic value of alternative uses. Payments and incentives for water conservation can increase water availability, just as pricing water at its full marginal cost can reduce demand. Functioning water markets can provide price signals for reallocation between different uses and also signals to guide conservation activities. Temporary trades have dominated experience with markets in developing and industrial countries alike. Experience with permanent transfers in Chile suggest that laissez-faire free markets without adequate consideration of third-party impacts will lead to adverse social and environmental consequences. Conversely, U.S. experience suggests that too much emphasis on water as a public and local resource is likely to greatly limit market activity, particularly permanent transfers. This will greatly constrain the achievement of economic efficiencies and ecosystem restoration. The Australian experience shows the potential of markets for reallocation to higher value uses. However, it demonstrates the need to be explicit about instream needs and to properly plan for the reintroduction of "unused" water that accompanies market development if environmental objectives are also to be met.

Key challenges in the development of payments for watershed service initiatives are to build capacity for place-based monitoring and assessment, to identify services in the context of the entire flow regime, to consider trade-offs and conflicts among multiple uses, and to make uncertainty explicit. Payment arrangements for ecosystem services provided by watersheds have been narrowly focused on the role of forests in the hydrological regime, they should instead be developed in the context of the entire flow regime, which would include consideration of the relative values of other kinds of land cover and land uses, such as wetlands, riparian areas, steep slopes, roads, and management practices. The value placed on watershed services will depend on stakeholder confidence in the effectiveness of proposed management actions for ensuring that the service continues to be delivered and that those who pay the costs will have access to the stream of future benefits. A precise determination of costs and benefits and their distribution presumes the ability to link actions and outcomes, so as to be able to demonstrate trade-offs. However, watershed processes are inherently variable and uncertain. Market mechanisms, on the other hand, tend to be more effective when uncertainty is low, because buyers like to know that they are getting what they pay for. Initiatives have often been based on myths and inappropriate generalizations about land and water relationships. While this may work in the short term, making uncertainty explicit is likely to be critical in managing buyer expectations and maintaining their cooperation in the long term.

A variety of public/private partnerships will have a role to play in financing water infrastructure for the provision of fresh water. There is a clear mismatch between the high social value of freshwater services and the resources that are being allocated to manage water. Insufficient funding to expand water infrastructure is one manifestation of this mismatch. Both inherent characteristics of the water sector (high fixed cost, low returns, long payback periods) as well as institutional problems (political interference, inadequate legal frameworks, poor management structures) explain the gap in funding infrastructure. A single-minded focus on large-scale privatization as a means of improving efficiency and cost-recovery has proven a double-edged political strategy—price hikes imposed by multinationals or acquisition of control over available resources have led to controversy and, in some cases, failure and withdrawal. To actually acquire "new" monies for investment, a fundamental change is required. At a national level, legal frameworks have to provide more certainty to the parties of long-term commitments. The water sector has to establish its priorities in a clear way and produce programs that include the definition of financing needs and sources. Finally, at the agency level, cost recovery must be improved and managerial and technical capacities enhanced.

New development of water infrastructure and technologies must observe best practices to avoid past problems and inequities; however, it is the reexamination and retrofitting/refurbishment of existing infrastructure that offers the most opportunity in the short and medium term. In regulated freshwater ecosystems, the optimal use of environmental flows will often require altered management of water infrastructure, supported by institutional arrangements across scales and actors. Once an environmental flow regime has been identified, management of freshwater ecosystems is likely to change. In highly modified and regulated systems, this may require decommissioning of dams or mitigating and altering dam operations and other water resource infrastructure, for example, managed flow releases. This predominantly requires an institutional response to facilitate these changes and may be accompanied by technological changes to retrofit infrastructure.

Table 7.1. Ecosystem Services Provided by Fresh Water and the Hydrologic Cycle. Many of the provisioning, regulatory, and cultural services can be enhanced through development of water resources (large-scale navigation can be increased by creating slackwater systems using dams); however, there are often off-setting losses or trade-offs between these service categories, such as loss of rapid transport downstream to locals or those seeking recreation.

Provisioning Services	Regulatory Services	Cultural Services
• Water (quantity and quality) for consumptive use (for drinking, domestic use, and agriculture and industrial use) • Water for nonconsumptive use (for generating power and transport/navigation) • Aquatic organisms for food and medicines	• Maintenance of water quality (natural filtration and water treatment) • Buffering of flood flows, erosion control through water/land interactions and flood control infrastructure	• Recreation (river rafting, kayaking, hiking, and fishing as a sport) • Tourism (river viewing) • Existence values (personal satisfaction from free-flowing rivers)

Supporting Services

- Role in nutrient cycling (role in maintenance of floodplain fertility), primary production
- Predator/prey relationships and ecosystem resilience

7.1 Human Well-being and Fresh Water

Ecosystem services are the benefits provided to people, both directly and indirectly, by ecosystems and biodiversity. In the Millennium Assessment, fresh water is a "provisioning" service as it refers to the human use of fresh water for domestic use, irrigation, power generation, and transportation. (See Table 7.1.) However, fresh water and the hydrological cycle also sustain inland water ecosystems, including rivers, lakes, and wetlands. These ecosystems provide cultural, regulating, and supporting services that contribute directly and indirectly to human well-being through recreation, scenic values, and maintenance of fisheries. Fresh water also plays a role in sustaining freshwater-dependent ecosystems such as mangroves, inter-tidal zones, and estuaries, which provide another set of services to local communities and tourists alike. This chapter explores how the trade-offs between these differing uses of fresh water and inland water systems can be balanced in the midst of increasing demand for all types of human benefit derived from fresh water.

7.1.1 Conditions, Trends, and Direct Drivers in Freshwater Services and Inland Water Ecosystems

In the past century, increasing human population and advancing levels of social and economic development have led to a rapid increase in the demand for freshwater provisioning services. In its natural state, fresh water varies considerably in terms of its availability in time and space. Water resources development—the construction of dams and irrigation channels, the construction of river embankments to improve navigation, drainage of wetlands for flood control, and the establishment of inter-basin connections and water transfers—has the aim of reregulating the natural hydrograph to meet human needs.

This has resulted in the replacement of naturally occurring and functioning systems with highly regulated and modified human-engineered systems. These "developed" systems have typically been designed solely for the satisfaction of the major human consumptive uses (irrigation or municipal and industrial use) or nonconsumptive use (hydropower and navigation).

These structural and capital-intensive responses—particularly large dams—have greatly augmented the natural availability of freshwater provisioning services. In the last 20 years alone, more than 2.4 billion people have gained access to water supply and more than 600 million have gained access to sanitation (World Water Commission 1999). At the same time, these supply responses have themselves become direct drivers of ecosystem degradation.

The impacts of water resource development are two-fold: less water remains in the ecosystem and the distribution and availability of the remaining water often has a different pattern from that present under natural conditions. It is estimated that the amount of water withdrawn from inland water systems has increased by at least 15 times over the past two centuries. (See MA *Current State and Trends,* Chapter 7, for a discussion of water withdrawals.) As a result, humans now control and use more than half of the continental runoff to which they have access. The impact of withdrawals, though, is not evenly spread and it is estimated that about 80% of the global population is living downstream of only 50% of Earth's renewable water supplies. (See MA *Current State and Trends,* Chapter 7.) Changes to the hydrograph and related physical, chemical, and biological processes have substantially degraded the condition of inland water ecosystems globally. (See MA *Current State and Trends,* Chapter 20.)

A related consequence of water resource development has been reduced water quality. Caused through the pollution of inland water ecosystems, this has occurred in parallel with the growth of urban, industrial, and agricultural systems. The major pollutants affecting water quality include nutrients, which drive eutrophication; heavy metals; nitrogen and sulphur based compounds, which cause acidification of freshwater ecosystems; organic compounds; suspended particles, both organic and inorganic; contaminants such as bacteria, protists, or amoebae; and salinity. According to the World Water Commission, more than half of the major rivers of the world are seriously polluted (WWC 1999). The presence of these pollutants depletes the capacity of rivers and associated inland and coastal ecosystems to provide clean water for social and economic uses.

Changes in the condition of freshwater and associated inland water ecosystems have also occurred at the hands of other direct drivers such as species introductions, land use change, and climate change. (See Table 7.2 and MA *Current State and Trends,* Chapters 7 and 20.)

7.1.2 Indirect Driving Forces

Most water-related problems, although caused by direct drivers such as water abstraction and pollution, are ultimately a product of indirect drivers. The development of water resources over the past century has been largely a result of the need to supply expanding populations with food, energy, and domestic and industrial water supplies and to facilitate opportunities for transport. Economic growth has further served to enhance the demand and consumption of freshwater services.

However, given the public as well as private good characteristics of fresh water, most water-related problems are ultimately a product of indirect drivers associated with the economic nature of fresh water in all its guises—and the manner in which this nature is accommodated or not by the institutional arrangements that govern the production, allocation, distribution, and consumption of freshwater services. The economic characteristics of fresh water, when combined with the dynamic nature of the hydrological cycle, present special challenges in the case of fresh water.

The potential for fresh water or ecosystems to have multiple uses, some of which will be private goods and others of which will either be perfect public goods or variations such as common pool or toll goods, creates this management challenge, as each type lends itself to a different management regime (Ostrom et al. 1993; Aylward and Fernández-González 1998). The market failure associated with public good characteristics suggests a need for mechanisms of social coordination in the form of institutional arrangements that can define, and adaptively manage, the level of provision and allocation of these goods and services that is desired by society.

Governance and the role of economic incentives are therefore critical indirect drivers with respect to balancing competing demands for freshwater. The inadequate governance associated with water resource development, particularly a single-minded, engineering-economic approach to the ecosystems services that inland water systems provide, has led to significant social and environmental impacts—impacts that have disproportionately affected the rural poor that rely on the natural functioning of inland water ecosystems (WCD 2000).

In the last two decades, increased attention has been paid to the importance of considering water as an economic commodity (Cosgrove and Rijsberman 2000). This has provoked considerable concern and controversy with respect to financing water infrastructure and water pricing, noticeably with regard to privati-

Table 7.2. Summary of Direct Drivers (Postel and Richter 2003)

Human Activity (Direct Driver)	Impact on Ecosystems	Services at Risk
Dam construction	alters timing and quantity of river flows. Water temperature, nutrient and sediment transport, delta replenishment, blocks fish migrations	provision of habitat for native species, recreational and commercial fisheries, maintenance of deltas and their economies, productivity of estuarine fisheries
Dike and levee construction	destroys hydrologic connection between river and floodplain habitat	habitat, sport and commercial fisheries, natural floodplain fertility, natural flood control
Diversions	depletes stream flow	habitat, sport and commercial fisheries, recreation, pollution dilution, hydropower, transportation
Draining of wetlands	eliminates key component of aquatic ecosystem	natural flood control, habitat for fish and waterfowl, recreation, natural water purification
Deforestation/land use	alters runoff patterns, inhibits natural recharge, fills water bodies with silt	water supply quality and quantity, fish and wildlife habitat, transportation, flood control
Release of polluted water effluents	diminishes water quality	water supply, habitat, commercial fisheries, recreation
Overharvesting	depletes species populations	sport and commercial fisheries, waterfowl, other biotic populations
Introduction of exotic species	eliminates native species, alters production and nutrient cycling	sport and commercial fisheries, waterfowl, water quality, fish and wildlife habitat, transportation
Release of metals and acid forming pollutants into the atmosphere	alters chemistry of rivers and lakes	habitat, fisheries, recreation, water quality
Emission of climate altering air pollutants	potential for changes in runoff patterns from increase in temperature and changes in rainfall	water supply, hydropower, transportation, fish and wildlife habitat, pollution dilution, recreation, fisheries, flood control

zation of municipal water supply, as well as with the application of market approaches, particularly with respect to the development of water markets and the use of payment systems for watershed services.

The discussion above highlights that water is in fact a resource that is often multifunctional and heterogeneous in nature. It is therefore not amenable to simple classification as either a public good or a private good. While water may be managed more successfully when its ''economic'' characteristics are recognized, due to its public good attributes, the solution will not be to treat it as a unidimensional commodity. Conversely, simply assuming fresh water is a public good in all contexts and uses is equally likely to lead to ruin. Rather, there is a need to respond to the inherent complexity of fresh water and work in an adaptable fashion toward site-specific solutions that accommodate the attributes and uses of fresh water in the local context.

7.1.3 Future Freshwater Challenges

While water resources development has increased the contribution of freshwater resources to human well-being, this is not to say that human needs for provisioning services are met today. Global figures indicate that 1.1 billion people do not have adequate access to good quality drinking water, 2.6 billion people lack access to sanitation, 800 million people do not have enough to eat, and 2 billion people do not have access to household electricity (UNESCO 2003; WHO and UNICEF 2004).

The International Food Policy Research Institute and the International Water Management Institute examined the impact of the world's growing human population on water and food (Rosegrant et al. 2002). With a global population of 8 billion people—a 2 billion increase—and a business-as-usual scenario, an overall increase in water withdrawals of 22% over 1995 levels is expected by 2025; this includes increases of 17% in the demand for water for irrigation, 20% in the demand for water for industry, and 70% in the demand for water by municipalities. Under a crisis scenario, a 37% increase in overall withdrawals is forecasted, while under a sustainability scenario, significantly less water—20%—is actually consumed in meeting future needs. Increased efficiency actually improves environmental flows and reliability of supply to agriculture.

The United Nations Millennium Development Goals set out ambitious targets with respect to human nutrition, education, and health, as well as for the environment. A target of halving the number of people who cannot access safe drinking water is the only explicit water-related MDG. Still the full range of freshwater ecosystem services will come into play—directly or indirectly—in achieving these goals (UN World Assessment Program 2003). The role of water in agriculture in meeting future demand and the MDGs is critical, and the ability to realize increased water productivity (getting more food for every drop of water) in irrigated and rain-fed agriculture will be of particular importance if water is to be freed up for ecosystems and other uses (Molden and Falkenmark 2003).

Meeting these provisioning needs will come at a significant cost. The report of the Camdessus Group to the Third World Water Forum in Kyoto, 2003, suggests that in developing countries, current spending on water services of $75 billion a year needs to be increased to about $180 billion if the water and sanitation MDGs are to be met (World Panel on Financing Water Infrastructure 2003). The food, water, and environment assessment currently being undertaken by the International Water Management Institute should yield additional information on the challenges and investments required with respect to agriculture and water (IWMI 2002).

Although needs and demands for provisioning services will continue to rise, it is increasingly accepted that any management response will need to deal with the potential for trade-off between the levels of different services provided by freshwater and associated inland water ecosystems. For example, the World Commission on Dams recently concluded that, although large dams have made an important contribution to development, this contribution has come at too high a price—referring to the high economic, social, and environmental costs (WCD 2000).

Efforts to ''mitigate'' the adverse impacts of traditional engineering responses to increasing supply are now the norm in most countries and, in many countries, efforts are now made to avoid these impacts in the first place by opting for other alternatives, including nonstructural approaches that dampen demand. At the same time, changing preferences in developed economies in favor of cultural and supporting services of inland water systems—such as tourism and recreation—imply that the balance of current effort is on ecosystem restoration rather than increasing provisioning services (for example, through new dams).

7.1.4 Optimizing Freshwater Ecosystem Services

Fresh water is a finite resource that cannot be distributed such that all the ecosystem services that it provides are maximized. This is the lesson of the environmental impacts observed across the world from water resource development. Ultimately, any development of water resources will involve a trade-off between provisioning, and the cultural, regulating and supporting services. In the past, the tendency was to sacrifice supporting, regulating, and cultural services in return for augmenting provisioning services. Increasing recognition of the consequences of such an approach has led to initiatives at all levels to address this issue and redress the balance.

However, current trends are likely to lead to a continued imbalance (see MA *Current State and Trends,* Chapters 7 and 20). Current trends to continue favoring provisioning services should reduce poverty. Nevertheless, due to the linkage between ecosystems and their cultural, regulatory, and supporting services, it is expected that poverty can only be reduced so far before feedback loops from ecosystem degradation will cascade back through these services, thereby reducing well-being, particularly for the poorest members of society. For example, in the Aral Sea, the benefits initially gained from increased agricultural productivity have been outweighed by the loss of fisheries and the impacts on human health, such as pulmonary diseases, from salt on the exposed seabed. Figure 7.1A shows the effect of these trends on freshwater ecosystem services that contribute to human well-being using a spider diagram.

An alternative is to strengthen the implementation of existing protective measures for the ecosystem. For example, full implementation of many conventions, laws, and rules would radically reverse ecosystem degradation and biodiversity loss. However, it is equally likely that full protection would greatly limit opportunities for meeting the continuing and growing needs for freshwater provisioning services with the probable consequence that poverty would increase. This scenario is shown in Figure 7.1B.

The alternative, of course, is to attempt to balance the services by optimizing across their full range. Figure 7.1C portrays such a generic scenario. Neither provisioning services nor the cultural, regulatory, and supporting services will themselves be maximized, but the overall effect would be to maximize welfare—subject to the constraint of targeting poverty reduction.

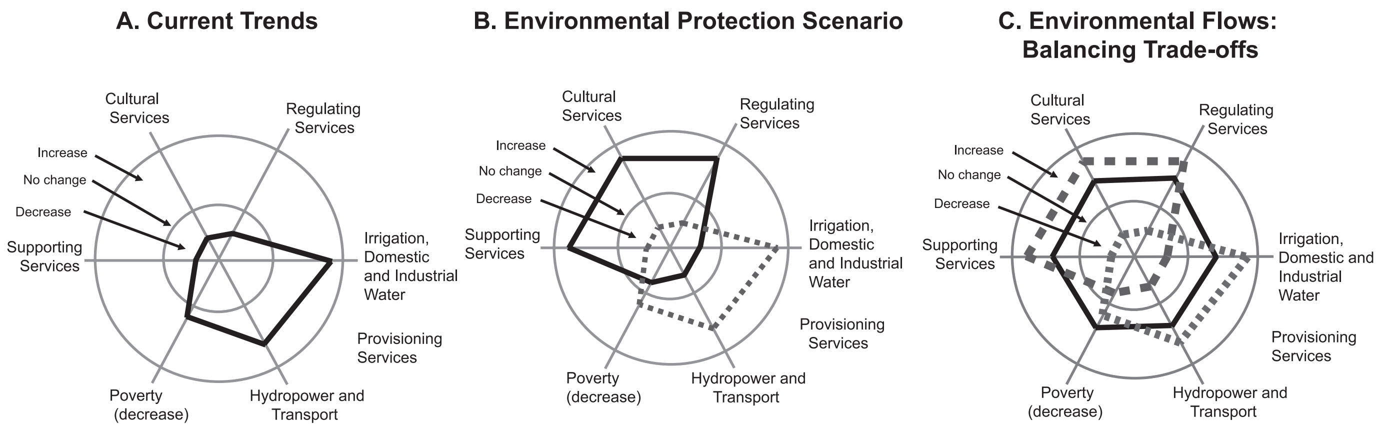

Figure 7.1. Balancing Trade-offs between Services and Impacts on Poverty

Clearly, with the challenges outlined above—both in terms of the demand and the need for appropriate institutional arrangements—the use of the term "optimizing" is very optimistic. In many contexts, efforts will be aimed at making the best of a difficult situation, striving for improvement and balance rather than optimization. Yet it is important to acknowledge, at least conceptually, that trade-offs among services present the possibility of optimization. While the responses selected to work in this direction will vary, it will be important to recall that the overall objective is to increase the overall productivity of all freshwater services and ecosystems.

One of the difficulties in assessing future needs is that there are many responses that are available. For example—as in the aforementioned IFPRI and IWMI forecasts—estimates of increases in the demand for water for agricultural purposes are highly variable due to differing assumptions about the productivity of water in agriculture (Molden and de Fraiture 2004). Raised to a higher level, the same concept applies. Inertia and difficulties in the governance of water is a significant problem, but if headway can be made, then there is at least the hope that existing technological and market approaches can reduce crisis estimates of the gap between supply and demand.

7.1.5 Selection of Responses for Assessment

With world population expected to stabilize in the middle of the next century, it is likely that the pressure on freshwater resources and inland water systems will never be greater than in the coming decades. In other words, it is critical that responses undertaken in the next 25–50 years accommodate increasing pressure for provisioning services, yet protect ecosystems, biodiversity, and their services. If provisioning services are consumed at the expense of ecosystems, the transition to a sustainable population will take place with a greatly reduced stock of natural capital with respect to freshwater services and inland water ecosystems.

A wide range of potential responses to the provision of fresh water and associated ecosystem services exist. Following on the typology presented in Chapter 2 of this volume, a rather exhaustive list of responses can be derived. (See Box 7.1.) This chapter distills responses from this broad list into those relating to governance and supply and demand management based on an identification of the key drivers that occur now and are expected in the future. A key message of this assessment is that underlying the selection of any responses must be recognition and consideration of the trade-offs implicit in selecting various response options.

The responses examined are outlined in Table 7.3. They include more traditional techniques, including supply-driven engineering responses and ways of applying these to yield more technically efficient, socially equitable, and environmentally responsible solutions. Alternatives are also examined that aim to increase the technical and economic efficiency of existing responses, rectifying inequities of past developments and restoring water quantity and quality of freshwater systems, as well as their diversity. The responses selected for assessment are intended to address a number of the critical future needs of policy- and decision-making as identified above and in the MA user needs analysis.

The responses were selected to illustrate different, but often complementary and mutually reinforcing, approaches. As a result, except for purposes of discussion, the responses are not always separate and discrete, but instead are often nested, one within the other or at different scales. Thus the choices made with respect to arrangements for property rights may determine the types of economic approaches that are used, which, in turn, may influence whether and what type of technology is ultimately applied in improving water resource management.

For example, improving the effectiveness of governance and developing appropriate institutional arrangements is an important response in itself, and is also an enabling condition for most, if not all, other responses. This is illustrated by market-based mechanisms, which would not work unless there are defined property rights and confidence that contractual agreements will be enforced. They may also rely on regulations that establish caps on water abstraction or pollutant emissions, and thereby create an incentive to use such approaches so as to reduce the costs of compliance.

While the limitation of responses that have proven inadequate in the past are covered, this is mostly to provide a context for the main focus, which is on what is being learned from new kinds of responses. These often require the development of new kinds of knowledge and other social capacities, and may present a whole different set of problems, not necessarily better or worse. For example, payments for watershed services have generated much enthusiasm, as they are seen as a more straightforward and direct approach to creating incentives for conservation that also helps to alleviate poverty, than integrated conservation and development projects, whose limitations are discussed extensively in Chapter 15; integrated responses often have numerous conditions that must be met for success. However, payments for watershed services, even if an improvement, present a different set of challenges and numerous unanswered questions as to their actual transaction costs.

We begin with issues of governance, which includes the challenges of determining ecosystem water requirements, rights, and

BOX 7.1

Response Options for Fresh Water and Related Services from Inland Water Ecosystems

Legal and regulatory interventions include:
- ownership and use rights at different administrative levels,
- regulation of pollution,
- regulation of environmental flows and artificial flood releases,
- legal agreements for river basin management, and
- regulations related to ecosystem and species conservation and preservation.

Economic interventions include:
- markets and trading systems for flow restoration and water quality improvements,
- payments for ecosystem rehabilitation,
- point source pollution standards and fines/fees, taxes,
- demand management through water pricing, and
- payments for watershed services.

Governance interventions include:
- participatory mechanisms (for example, watershed/catchment councils and farmer-based irrigation management systems),
- river basin organizations (international or regional scale),
- integrated water resource management and basin planning,
- private sector participation, and
- institutional capacity building (for example, for regulatory agencies).

Technological interventions include:
- water infrastructure projects (such as dams, dikes, water treatment and sanitation plants, desalinization),
- soil and water conservation technologies (such as physical and vegetative measures for soil and water conservation),
- end-use and transmission efficiency options (such as drip irrigation and canal lining/piping),
- demand management/technologies for higher end-use efficiency (such as low-flow showerheads, energy conservation programs/incentives), and
- research into water-saving technologies and breeding crops for drought tolerance.

Social, cultural, and educational interventions include:
- environmental education and awareness,
- making explicit the value of non-provisioning water ecosystem services, and
- research into land–water interactions in a watershed context.

responsibilities in the provision of fresh water, increasing the effectiveness of public participation, river basin organizations, and the role of regulation. Experience with the use of economic incentives for supply and demand management is then examined with particular reference to reallocating fresh water and conserving watershed hydrologic function, as well as mobilizing financing for the development of water supply infrastructure through public-private partnerships. Finally, supply infrastructure and technologies are covered briefly, with assessments of two in particular: a well-established response (large dams) and a new response (wetland restoration and mitigation). Other chapters address responses to issues of climate change (see Chapter 13) and issues of increasing the productivity of food production (Chapter 6).

7.2 Governance: Institutions for Managing Shared Waters

The objective of achieving a reasonable balance between the provisioning of fresh water and maintaining the underlying ecosystem processes that support fresh water and related ecosystem services implies a fundamental shift in water management approaches. This shift is from an emphasis on the continued development of new sources of water supply toward the reallocation of water among various uses and management of demand; recovery of costs of conservation management and research activities; and operations and maintenance of systems for delivery of water and sanitation services. This new emphasis is consistent with the integrated water resources management approach defined by the

Table 7.3. Response Options for Optimizing Human Well-being from Freshwater and Associated Inland Water Ecosystems

Governance	Supply Management	Demand Management
Defining ecosystem water requirements Property rights Participation in decision-making River basin organizations and transboundary management Regulatory	Economic incentives for reallocation and new supply • partnerships and financing • water markets • cap and trade systems • payments for watershed services Infrastructure • large dams • levees • locks and canals Technologies • wetland restoration • agricultural water conservation • desalinization • rainwater harvesting	Economic incentives for consumers • water pricing • payments and subsidies for on-farm and household water conservation Water conservation technologies • on-farm water efficiency and management improvements • municipal and industrial water measurement and savings devices

Global Water Partnership (2000), which talks of managing water so as to advance a country's social and economic development goals in ways that do not compromise the sustainability of vital ecosystems, by taking into account the broader ramifications of sectoral actions.

Integrated water resources management is a challenge of governance as well as of economics and of obtaining adequate scientific knowledge, as it requires making decisions regarding the trade-offs among the multiple and often conflicting uses and interests that often accompany changes in policy objectives and management practices. Implementation also requires the development of appropriate and effective institutional arrangements that provide stakeholders with the incentive to cooperate and comply with management plans.

Because of the vital importance of water to human society, problems in water management also reflect more general weaknesses in governance, and provide a point of departure for broader policy reforms, and vice versa. This is explicitly recognized, for example, in the European Water Framework Directive, which requires harmonization with policies in other sectors. The Water Framework Directive Common Implementation Strategy gives priority to achieving consistency of agriculture, transport, energy, internal markets, development, fisheries, and marine policies, with WFD water policy objectives. Conversely, events such as the fall of apartheid in South Africa made possible major reforms in water policy. In the Danube Basin, cooperation on basin-wide management issues only became possible with the end of the Cold War, where efforts to achieve consistency with the Water Framework Directory are intertwined with the development of democratic institutions.

As is pointed out in the World Water Development Report of the United Nations World Water Assessment Program (UNESCO 2003), there has been a significant expansion, over the past 20 years, of international programs and institutions that pertain to fresh water, beginning with Agenda 21, and these include:

- new national water laws,
- international agreements,
- the formation of river basin organizations,
- the establishment of the World Water Council, which sponsored the two World Water Forums and an independent World Water Commission to produce a World Water Vision, and
- establishment of the Global Water Partnership to form local and regional partnerships.

While these bodies have contributed to broader awareness of water problems and solutions, there is also a general acknowledgement that there is a wide gap between formal policies and actual practices. For example, as stated in a report regarding watershed management in Peru, while there has been an evolution of the concept of watershed management, toward more integrated and participatory approaches, which encompass social as well as environmental aspects over larger spatial areas, these are taken up more easily in international discourse than in actual watershed management practices (Bellido 2003). Continued effort to support more effective on-the-ground implementation is always useful, particularly if placed within a structured framework for learning and adaptive management.

Perhaps the most significant methodological challenge is to develop the capacity for a place-based approach to assessment, which is necessary to identify ecosystem functions that support the provision of valued ecosystem services in a specific context, and to select feasible and appropriate institutional arrangements. This requires consideration of the entire flow regime, which provides a framework for considering the full range of interests in the context of the specific ecosystem functions that support them, and for identifying trade-offs, including uncertainties. It can be regarded also as an institutional challenge in that impacts on livelihoods and other aspects of well-being are easily overlooked in the absence of effective participation and recognition of the rights of the various stakeholders.

The responses to the challenge of developing effective institutions addressed in this section include (1) determining ecosystem water requirements, (2) determining rights to freshwater services and responsibility for their provision, (3) increasing the effectiveness of public participation in decision-making, (4) developing river basin organizations, and (5) regulatory responses.

7.2.1 Determining Ecosystem Water Requirements

In middle-income developing and industrial countries, where basic needs for food, water, power, and transport have largely been met by past water resource development, attention is refocusing on the regulating, supporting, and cultural services that well-functioning inland water ecosystems can provide. Given the history of development of land and water resources, this often requires efforts to restore the natural function of watersheds, rivers, and wetlands. There has also been a push in some developing countries to consider ecosystem water requirements before installing water resource infrastructure and associated allocation systems. (See Box 7.2.)

The desire to determine ecosystem water requirements can be driven by anecdotal observations of a decline in certain ecosystem

BOX 7.2

The Lesotho Highlands Water Project (Brown and King 2003)

The Lesotho Highlands Water Project is aimed at regulating water resources across the headwaters of the Senqu River system in Lesotho through the establishment of large dams and weirs. River regulation is considered economically valuable for Lesotho and downstream in South Africa's Vaal region. The Lesotho Highlands Water Commission, a bi-national body, oversees the project.

In 1986, a treaty was signed to initiate the project, which included provision for renegotiation after 12 years. Renegotiation of the terms of the project were delayed in 1998 so that an assessment could be undertaken to predict the biophysical, sociological, and economic responses of major infrastructure works, as well as mitigation and compensation costs for affected subsistence groups that use the river. This was done using the Downstream Response to Imposed Flow Transformations (DRIFT) approach by examining responses under different flow regulation scenarios.

The major potential water conflict issues were the impact of regulation on the 39,000 subsistence users downstream of the dam sites and the potential impact on ecosystem services such as ecotourism. In general, it was predicted that the more water released downstream of storage, the less the change in river condition and associated socioeconomic impacts to riparian people, but the greater the impact on system yield, and hence, potential for revenue through development opportunities.

The result of the application of DRIFT was that flow releases from some of the major dams and weirs have been altered and the design and operation of news dams will be modified (as at the Mohale Dam), particularly to facilitate environmental release in the form of seasonal releases or small floods.

features and functions or by the results of applied analysis spawned by observation. A change in condition may be detected by local communities (for example, the decline in catch size of native fish and the change in vegetation composition on floodplains) or through government-sponsored surveillance monitoring programs conducted as a legislative requirement. Determining ecosystem water requirements may also be undertaken for undeveloped water resources, although this is uncommon given that most available water resources across the globe are already developed to some degree.

Many of the services provided by inland water ecosystems are supported by natural variability in spatial and temporal patterns in the distribution, abundance, and quality of water and the interaction between the basin, climate, geology, topography, and vegetation of which they are a product (Poff et al. 1997; King and Brown 2003). Other processes central to ecosystem functioning are also sustained by flow such as sediment and nutrient transport (Whiting 2002). The development of water resources, while often benefiting provisioning services, has modified these patterns and deleteriously impacted regulating, supporting, and cultural services. (See MA *Current State and Trends,* Chapters 8 and 23.)

Components of the flow regime are the natural patterns of variation in the quantity and timing of the flow of a river, including the natural disturbances associated with these flow patterns (Poff et al. 1997). The need to maintain adequate quantity and quality of water in inland water ecosystems is captured by the notion of ecosystem water requirements and management, which are referred to in much of the literature on river ecology, where this concept has the greatest currency as "environmental flows."

Put simply and based on concepts developed for rivers, environmental flows are the water that is left in a river ecosystem, or released into it, for the specific purpose of managing the condition of that ecosystem (Brown and King 2003). Applied more liberally, ecosystem water requirements can be taken to mean the water left in an inland water ecosystem, or released into it, for the specific purpose of meeting objectives with regard to the balance of services provided by that ecosystem. This could apply equally to coastal systems dependent on freshwater supply. While the definition of ecosystem water requirements provided here is broad, their determination has been developed almost entirely for riverine ecosystems.

A failure to determine and allocate water for ecosystems may not only result in the loss of the natural functions of the parent system, as has been observed following water resource development on the Colorado, Ganges, Nile, and Yellow rivers (Postel 2000), but also cause degradation of dependent ecosystems, as for example in the loss of biotic production in the Sea of Cortez at the mouth of the Colorado River (Baron et al. 2002). These and other examples from across the world have led to the recognition that the ecological sustainability of inland water ecosystems is threatened by the hydrologic alterations carried out by humans (Poff et al. 2003).

Defining ecosystem water requirements targets the direct drivers of resource consumption in the form of water abstraction and the reregulation of the timing of flows, the objective being to determine the quantity, quality, and timing of water that should remain "instream" or within an ecosystem for the variety of regulatory, supporting, and cultural services it sustains. Given the site-specific nature of what might constitute ecosystem water requirements, a wide range of approaches exist for implementing these flows. (See Box 7.3.)

The institutions involved in undertaking ecosystem water requirement determinations are usually those that have the required governance and technical capacity. These are principally governments looking to inform natural resource management policy-making, or not-for-profit groups wanting to inform advocacy positions. The process will typically occur at a catchment scale, but will require the involvement of actors from a local through to a regional and national scale.

There are essentially two steps in most techniques used to define ecosystem water requirements: the setting of objectives and the definition of the water (quantity and quality) required to meet those objectives supported by technical advice on the condition of the river system under various flow regimes (Acreman and King 2003).

BOX 7.3

Approaches to Implementing Environmental Flow Regimes

In rivers and streams, restrictive flow management—where abstractions and diversions are controlled—can be used to achieve an environmental flow, for example, the Murray-Darling Basin cap (Acreman and King 2003, see Box 7.5). Depending on abstraction levels and the identified needs of the system, water recovery may also be required, which can be achieved through reducing allocations for consumptive uses such as irrigation (Postel 2000). In this instance, active flow management may be necessary and involve management of "environmental water" through practices such as managed floods. This method is gaining support among river managers globally and has been implemented in rivers such as the Senegal, Phongolo, and Kafue in Africa (Acreman et al. 2000), Colorado in the United States (Webb et al. 1999), and the Murray in Australia (Siebentritt et al. 2004).

7.2.1.1 Defining Ecosystem Condition: A Balance between Ecosystem Services

Maximizing human well-being in freshwater dependent ecosystems requires a decision involving all the stakeholders on the desired condition of an ecosystem (Poff et al. 2003). Such a decision needs to balance human aspirations for water resource development for provisioning services, with the indirect water requirements needed for other ecosystem services (Acreman 2001).

Historically, an indirect and unconscious decision was made on the amount of water required to sustain an ecosystem. For example, damming a river and diverting water for irrigation prior to the development of our current understanding of flow ecology relationships effectively represented a decision to not supply water to other ecosystem services. The process of determining ecosystem water requirements aims to make a conscious and intentional decision on how water is distributed among different services.

Invariably a variety of views exist among stakeholders on what the desired level of ecosystem services should be and the trade-offs that are acceptable in improving human well-being. For example, reduced water for abstraction may mean less irrigation, but this may provide more water to achieve desired levels of fishing, boating, ecosystem resilience, and scenic values. Setting this balance has sparked debates across the world. Decision-makers responsible for water allocation may seek the minimum flow that must remain in a river to maintain the environment. This approach encompasses the notion of resilience, which refers to the thresholds within which changes to natural flow regimes can be adjusted. However, such thresholds below which the ecosystem abruptly degrades are very illusive and may not exist in reality (Acreman in press).

7.2.1.2 Methodologies for Defining Ecosystem Water Requirements

Recognition of the need for adequate quantity, quality, and timing of flows in river systems has led to the development of over

200 methodologies to determine ecosystem water requirements (Tharme 2001). Many of these originated in the United States, South Africa, and Australia (Postel and Richter 2003). All make the assumption that aspects of flow are critical for maintaining certain ecosystem features.

Methods vary from those with a focus on defining minimum instream flow requirements for a few high-profile species to more recent efforts to describe the full spectrum of flow characteristics needed to maintain the ecosystem condition (that is, the flow regime), including low flows, higher flows, large floods, specific rates of rise and fall in water levels, and annual variability in discharge (Poff et al. 1997; Postel and Richter 2003). In this context, the flow regime provides a framework for considering the full range of interests in the context of the specific ecosystem functions that support them, and for identifying and quantifying the trade-offs associated with various scenarios, both in biophysical and socioeconomic terms.

Despite the large number of methods, each of which is unique in some regard, many methods are broadly similar (Dunbar et al. 1997) and aim to draw links between hydrological and hydraulic features with ecological responses, whether of individual species and populations, or their habitats. (See Table 7.4.) Further differentiation can be made between bottom-up approaches that build a flow from a base of mandatory low flows and then add freshets and floods (Arthington et al. 1998) and top-down down methodologies that determine the maximum acceptable departure from natural flow conditions (Brizga 1998).

The current trend is toward more holistic approaches that incorporate a number of flow assessment methods and examine all aspects of the ecosystem. Application of these methods requires multiskilled teams, including hydraulic engineers, hydrologists, ecologists, and geomorphologists, as well as those with skills from the social sciences (Postel and Richter 2003). Where expertise is minimal, more interactive approaches may be considered that draw on local and traditional knowledge.

Table 7.4. Advantages and Disadvantages of Different Methods and Characteristics of Setting Environmental Flows

Method Type	Advantages	Disadvantages
Look-up table	inexpensive, rapid to use once calculated	not site-specific hydrological indices are not valid ecologically ecological indices need region-specific data to be calculated
Desk top	site-specific limited new data collection	long time series required no explicit use of ecological data ecological data too time-consuming to collect
Functional analysis	flexible, robust, more focused on whole ecosystem	expensive to collect all relevant data and to employ wide range of experts consensus of experts may not be achieved
Habitat modeling	replicable, predictive	expensive to collect hydraulic and ecological data

One example of such an approach is the Downstream Response to Imposed Flow Transformation framework (see Box 7.4), which can inform a decision on the desired condition of an ecosystem through assessment of scenarios. The DRIFT framework differs from other approaches such as the Instream Flow Incremental Methodology and the Catchment Abstraction Management Strategy in that it explicitly considers the socioeconomic implications of different scenarios (Acreman and King 2003).

Public participation to allow consideration of stakeholder views is central to the determination of ecosystem water requirements and was incorporated into the Murray-Darling Basin Commission's "The Living Murray" initiative (Murray Darling Basin Commission 2003). This involved a coordinated process of community engagement combined with socioeconomic and biophysical assessments to determine the desired condition of the River Murray, expressed as ecological objectives and outcomes for "significant ecological assets" from across the river system. (See Box 7.5.) In November 2003, this led to a decision to take a "first step" in recovering up to 500 gigaliters of water to support meeting these objectives.

Definition of the stakeholders to be included in assessments of water requirements will influence greatly the effectiveness of public participation efforts, and satisfaction of stakeholders with the outcomes of this process. There are arguments to suggest that in general (inter)national non-users should have a role in regional/local decision making, especially when important, non-replaceable features of an ecosystem are involved. Ultimately the solution to this issue is a definition of the boundary conditions and the degrees of freedom for debate among stakeholders.

7.2.1.3 Effectiveness

A common conclusion of practitioners in this field is that there is no single best method for determining ecosystem water requirements. Instead, application of flow assessment methodologies should be tailored to the unique set of ecological, water develop-

BOX 7.4

DRIFT Flow Assessment Methodology

The DRIFT (Downstream Response to Imposed Flow Transformations) methodology is an interactive and holistic approach for advising on environmental flows for rivers targeted for water-management activities. The methodology employs experienced scientists from a range of biophysical disciplines and, where there are subsistence users of the river, engages a number of socioeconomic disciplines. It produces a set of flow-related scenarios for water managers to consider, which describe a modified flow regime; the resulting condition of the river or species; the effect on water resource availability for off-stream users; and the social and economic costs and benefits. The process involves one or more multidisciplinary workshops, attended by a range of affected stakeholders to develop agreed biophysical and socioeconomic scenarios.

The development of scenarios requires an assessment of biophysical, social, and economic data and may draw on results from other predictive models that assess the responses of specific biota to flow (as in the physical habitat simulation model or PHABSIM). DRIFT should run parallel with two other exercises that are external to it: a macroeconomic assessment of the wider implications of each scenario, and a public participation process, whereby people other than subsistence users can indicate the level of acceptability of each scenario.

BOX 7.5

The Murray-Darling Basin Cap

(Murray-Darling Basin Commission 2003)

In June 1993, the Murray-Darling Basin Ministerial Council directed that a study be undertaken into the issue of altered flows and their consequences for the rivers of the Murray-Darling Basin. This led to an audit of water use, which confirmed increasing levels of diversions and associated declines in river health. In response, the Council introduced an interim cap on water diversions in the Basin in 1995. This was established as a permanent cap on July 1, 1997. In agreeing to the implementation of the cap, the Council essentially made a decision about the balance between the social and economic benefits derived from development of the Basin's water resource and the water needs of the riverine ecosystem. Implementation of the cap is the responsibility of the member states of the Murray-Darling Basin Initiative. Water audit monitoring reports are prepared annually to insure state water use is consistent with the cap and an independent audit group reviews progress on its implementation.

ment, and socioeconomic conditions in a river basin (Postel and Richter 2003). This is because of the variation in enabling conditions and binding constraints between each method and the prevailing conditions in a river basin.

The major issues when applying a methodology are the data required, technical capacity, funding, and time available to complete assessments (Arthington et al. 1998; Tharme 2001). The current shift toward more holistic methods brings with it a greater reliance on detailed ecological, hydrological, and hydraulic data. In the absence of hydrological data, computerized simulation models may be used to synthesize estimates of natural flow (Postel and Richter 2003). However, where the link between ecology and flow is unknown, predictions are more likely to call on expert opinion. Reliance on expert opinion is encapsulated in the Expert Panel Approach, which has been used widely across the eastern states of Australia with considerable success (Brown and King 2003).

The need for expert opinion highlights the often imperfect knowledge of the relationships between the ecology and hydrology of an ecosystem, perhaps the greatest source of uncertainty in determining ecosystem water requirements. This uncertainty will be less where the focus is on a single species or ecosystem component; but where a range of ecological and flow relationships are integrated, uncertainty will rise.

Various assessment techniques embrace uncertainty within their design. In the Lesotho Highlands Water Project, uncertainty was managed through the use of severity rankings, enabling scientists to indicate within a range how great each described change would be; where uncertainty surrounded predictions, the range of severity ranking was expanded (King and Brown 2003). Another method of expressing certainty in data quality was used in the Murray Flow Assessment Tool (based on an Environmental Flows Decision Support System; see Jones et al. 2003), whereby the quality and source of ecological data used for the assessment were recorded and open to interrogation by all users.

Time and money also influence the choice of method and approach. Expert panel methods are often rapid to carry out (Brown and King 2003), and because little to no additional data collection is required, cost is largely a function of the time of the experts. In contrast, more interactive approaches that require data collection across ecological, hydrological, and socioeconomic disciplines require more time and money. For example, an application of DRIFT could take up to three years (as in the Lesotho Highlands Water Project; see King and Brown 2003) and cost at least $1 million.

More recent environmental flow assessments have involved a broader array of stakeholders, including greater representation of community representatives and those who possess local or traditional knowledge—often these are the people who may be most directly impacted upon by upstream developments such as dam construction and water abstraction. Therefore, another binding constraint on the success of determining ecosystem water requirements is the extent to which public participation is embraced by local, regional, and national governing bodies.

In many countries where ecosystem water requirements are determined, a strong and effective regulatory framework and capacity for environmental protection exists. This framework serves to insure that additional ''development'' takes account of social and environmental issues (that is, through environmental impact assessment), can be used by activists to halt projects that threaten scarce environmental resources (for example, endangered species legislation), and provides for environmental standards and redress for specific environmental pollutants (for example, clean water legislation). Such a regulatory framework may be effective for restraining the environmental excesses of development; however it is often not sufficient to provide for a proactive response to restore degraded systems.

The effectiveness of the process for determining environmental flow regimes is influenced by the ability of practitioners to choose the right method following consideration of the enabling conditions and binding constraints for each application. Hence, this part of the process is effective provided an appropriate approach and mix of methods is adopted. The real test of the effectiveness of determining ecosystem water requirements lies in their allocation or implementation and the ability of the resulting flow or water regime to achieve its predefined objectives.

Where objectives are simple, such as minimum stream flows, they may be easy to measure and achieve. However, where the objective is to maintain a desired condition for a number of ecosystem components, the task is more difficult. In many instances, ecological response will be slow. Mitsch and Wilson (1996) suggest that it may take 15–20 years before monitoring can reveal the success (or lack of it) of large-scale wetland restoration projects. Determining the success of environmental flow regimes against economic, social, and environmental goals at the scale of the ecosystem, river basin, or even catchment may take considerably longer.

The success of changed river management to meet aspects of ecosystem objectives can, however, be judged somewhat by responses measured over the short term, as for example in reforming the sand bars in the Colorado River in the United States (Webb et al. 1999) and the expansion of the native perennial grasses on the Murray floodplain in Australia (Siebentritt et al. 2004) and on the Logone floodplain in Africa (Scholte et al. 2000). These responses, although not evidence of meeting objectives for the entire ecosystem, do indicate that responses are possible and consistent with broader objectives.

In the absence of the desired response, care needs to be taken in differentiating between a failed flow regime (the expression of a certain combination of ecosystem water requirements), a delayed response, and the presence of a system that is not capable of responding. The latter may occur if the system has been too severely degraded or if other factors are having a mitigating effect. With regard to the latter, patterns of water quantity and quality are rarely the sole drivers of degradation. Others include habitat modification, reduced longitudinal and lateral connectivity, salinization, and eutrophication. As such, the allocation of ecosystem water requirements should form part of wider river basin management strategies, of which they may be a cornerstone.

7.2.1.4 Findings and Conclusions

There is growing support for the conclusion that there is no single best method for determining ecosystem water requirements (Acreman and King 2003). Instead, selection needs to be informed by site-specific factors that match a flow assessment technique with the enabling conditions and binding constraints relevant to a particular location/site and application.

With respect to success thus far with determining ecosystem water requirements, where the objectives set are purely hydrological and relate to setting minimum flows, there are numerous examples of success. However, there would seem to be few examples of the effectiveness of the more recent trend toward holistic approaches that set objectives for a range of features of the flow regime and the ecosystem, and social responses to ecosystem change. This is largely a function of the long lead times for ecosystem-scale response.

Through incorporation of an adaptive management approach, the effectiveness of a flow regime designed to meet certain ecosystem objectives may be improved through time. Adaptive management is an integrated response option that provides a way to build on a base of imperfect knowledge. Active flow management through the use of environmental water ''allocations'' provides an ideal opportunity to learn more about the functioning of freshwater ecosystems and their water requirements. Such events can be treated as experiments to test management hypotheses.

7.2.2 Rights to Freshwater Services and Responsibilities for their Provision

As discussed above, freshwater services may span the range from public to private good. Due to their public good attributes, the value placed on freshwater services by actual or potential beneficiaries depends not only on demand, but also on whether they have confidence in the effectiveness of proposed management actions and institutional arrangements needed to insure that the service is actually delivered, and that they will have access to the stream of benefits. Access to benefits is determined by property rights, which refer not only to private individual rights, but are defined more broadly as ''the capacity to call upon the collective to stand behind one's claim to a benefit stream'' (Bromley 1989), and may include common property rights of communities and public as well as private forms (Ostrom 1990). Absence of enforceable rights is defined as a condition of ''open access.''

Rather than a pure concept of ''ownership,'' this broader concept of property refers to overlapping bundles of rights held by individuals, groups, or the state, which will depend on the characteristics of the resource and on valued uses. For example, some individuals may have use rights to bathe in a river or water their animals; a water users' association may have control or decision-making rights to divert some of the water (management) or exclude others from using it; and the state may claim the alienation rights to transfer the water.

Although the state may define property rights over these resources, in fact, there are multiple sources of land and water rights that need to be considered, including those that derive from:

- international law or treaties (the Ramsar Convention, for example, which affects rights and uses of wetlands);
- state (or statutory) law, which may in itself have many different rights defined for various uses and users;
- religious law and principles, such as Islamic principles of the ''right to thirst'' for humans and animals;
- regulations of particular development projects; and
- customary law and local norms.

Each type of property right is only as strong as the institution that stands behind it. While in some cases, the government has a very strong influence, it does not operate alone. In many cases, customary or religious law may be more influential. Because of this complexity, changes in state law alone do not automatically translate into changes in property rights on the ground (Meinzen-Dick and Pradhan 2002).

Instead, as the above suggests, changes in property rights generally occur as a result of contested processes between different kinds of overlapping and competing claims. Outcomes will inevitably depend on the political and economic power of those who stand to gain or lose, on scientific and technological advances that reduce transaction costs of controlling access to the service, the ability to transfer rights, and changes in social values as new kinds of problems emerge that threaten future provision of the service. For example, in the United States, riparian rights, which limited uses of water to levels that did not impair the natural flow (which was valued for supporting the operation of power mills), was replaced by rights to reasonable use needed to enable the higher consumptive uses required for irrigation, municipal, and industrial uses, thereby curtailing rights to natural flows and often to aquatic resources that rely on them (Sax 1993; Tarlock 2000). Because of new social goals of protecting the ecosystem, rights are again being contested and redefined (Fahrenthold 2004; Santopietro and Shabman 1990).

Depending on the outcome, this process of making and contesting claims on freshwater services can lead to the renegotiation of new kinds of rights and responsibilities in which the uses of land, water, and other natural resources are limited to those uses that do not impair service provision. They may also help to insure that those who pay the costs of management practices have access to their benefits through various forms of compensation. Without some form of entitlement, land users will not be in a position to enter into contractual agreements regarding land uses and management practices, nor will they have access to benefits from investments made in such practices, or be in a position to trade rights to fresh water for other goods and services (Tognetti 2001; Swallow et al. 2001).

The rise in values of these services may further disadvantage those who lack rights by leading to regulatory restrictions on their uses of water and land, and sometimes to their displacement, without any corresponding access to benefits. For example, a case study in Thailand suggests that dry season flows have diminished primarily because of a dramatic increase, both downstream and upstream, in dry season cultivation and irrigation of soybeans by those who own paddy fields. However, the focus of regulation intended to address the problem has been on the more vulnerable farmers, who are dependent on rain-fed slopes in areas where significant forest cover remains, who have the least significant impacts on hydrology, and who are regarded as guardians of resources rather than as legitimate users (Walker 2003). In many cases, protection of watersheds has been used by governments as justification for centralized control of land and water resources and regulatory conservation policies that exclude upland communities. This approach tends to be resisted by local populations using various means such as fire, which further reduce the effectiveness of regulatory approaches (Swallow et al. 2001; Blaikie and Muldavin 2004).

A key issue in defining rights and responsibilities is the extent to which land users should be compensated for the costs of management practices by those who directly benefit from them, and the extent to which such practices should be required through regulations, based simply on a responsibility not to pollute and not to harm others. In the former case, land users are paid for the costs of conservation management practices needed to insure continued provision of valued freshwater services, in addition to marketable agricultural and timber products. In the latter, to the extent there is compliance, the ability of farmers to stay in business will depend on the extent to which they are able to pass their costs on to consumers in the form of prices.

7.2.2.1 Effectiveness

The process of changing or redefining rights and responsibilities, so as to reflect new values placed on freshwater services, is primarily constrained by transaction costs. These are costs associated with the development and enforcement of rules or institutional arrangements that can effectively control access to specific services, monitoring the conditions of watershed processes that support the production of services, and the resolution of conflicts between new and existing multiple uses and values, as services become scarce.

The costs to develop and enforce rules, and monitor watershed processes, are largely related to the site-specific biophysical characteristics of the hydrological cycle as well as to those of the users and their social and economic context. These have implications for the kinds of arrangements that will be feasible and effective, the information needed to support management decisions, and the technical and institutional capacities that may need to be developed. For example, because of the relatively large size of upper watershed areas, conservation management practices may be necessary over large areas, well beyond the level of individual plots, before a significant change can be detected in the provision of services. It will also be necessary to target marginal and unproductive areas that contribute disproportionately to off-site impacts—steep slopes, gullies at the base of escarpments, river banks, forest margins, and paths and roads, for example. Given that these areas provide little if any return to individual landowners on investments in conservation practices, individual private property rights generally do not provide a sufficient incentive for landowners to make such investments. In the absence of a set of enforceable rules, such areas are, in effect, ''open access'' areas (Swallow et al. 2001). Emphasis on the registration of formal private rights also carries risks of overlooking informal rights found in existing practices and social norms for managing the water and land uses by which it is affected, of placing at a disadvantage those with less education and who have fewer social connections, and of increasing uncertainty by creating new rules without developing the capacity and willingness to enforce them (Meinzen-Dick and Bruns 2003).

A key factor in overcoming the above constraints and generally reducing transaction costs is to strengthen forums for negotiation of rights and responsibilities, which, particularly at smaller scales, are better suited than rigid legal frameworks for creating access rules that are responsive to site-specific and constantly changing conditions (Meinzen-Dick and Bruns 2003). Given that there are normally large differences in the relative power of various stakeholders, collective action may also be necessary just to enable marginalized stakeholders to obtain rights or even recognition of existing informal rights. By providing a space for regular patterns of interaction, such forums can also facilitate the development of trust—the *social capital* that enables stakeholders to collaborate and engage in the collective action needed for monitoring and stewarding large and marginal upper watershed areas. This may take the form of user associations and watershed councils that can negotiate on behalf of numerous users of water and land, as well as provide technical assistance in the preparation and implementation of joint management plans. This process may gain momentum from broader social and technological changes and fom effective decentralization of resource management; these are further discussed in subsequent sections.

A promising approach that has been identified in Indonesia and the Philippines is where upland populations are granted tenure that is conditional upon compliance with land management plans and co-management agreements (Swallow et al. 2001; Rosales 2003). In Indonesia, this approach only became possible with the fall of the Suharto regime, which enabled local populations to voice demands for change in the property rights regime and to resist coercion by the forestry department and commercial interests. The result is a current trend toward decentralization in which management rights are being granted in exchange for adhering to agreed upon management plans (Swallow et al. 2001).

A general area of conflict that will undoubtedly have implications for the outcomes of future initiatives is between efforts to recover costs of both delivery and conservation of fresh water through pricing and privatization, and the recognition of water as a more universal kind of human right, consistent with objectives of poverty alleviation. Both objectives, for example, are found in the fourth Dublin Principle, which recognizes water as an economic good, but states also that ''within this principle, it is vital to recognize first the basic right of all human beings to have access to clean water and sanitation at an affordable price.'' This dilemma is embodied also in the South African constitution and water law. Whether or not water pricing and privatization have enabled governments to expand services to poor and previously unserved populations addressed below. However, the approach is severely constrained by the ability to pay, and has been actively resisted through protests in a number of countries, including South Africa and several Latin American countries.

Among the better known was a protest in Cochabamba, Bolivia, where water privatization was brought to a halt after it led to monthly water bills that were double or triple what they had been previously and that, for many, amounted to 20% of their salaries (Rothfeder 2003). Full-cost pricing was also the subject of controversy during the development of the EU Water Framework Directive, which, in its final form, only required that environmental costs be taken into account in determining water prices (Kaika and Page 2002). In South Africa, this issue was addressed by designing price structures that distinguish between different uses so as to reflect policy priorities of providing for basic human and environmental needs. However, as the case of South Africa shows, the major challenge lies in implementation. (See Box 7.6.)

7.2.2.2 Findings and Conclusions

The legitimacy of various conflicting claims to freshwater services will depend on the outcome of negotiation and conflict resolution. Establishing such a process is central to sustainable management of freshwater services because it allows for the more flexible and adaptive responses needed to cope with the complexity and uncertainty of watershed processes than formal and rigid legal frameworks.

Both public and private ownership of fresh water, and also of the land resources associated with its provision, have largely failed to create adequate incentives for provision of services associated with it because upland communities have generally been excluded from access to benefits, particularly when they lack tenure security and have resisted regulations regarded as unfair. Public and

BOX 7.6

Water as a Human Right and an Economic Good in South Africa

The constitution of South Africa guarantees access to sufficient water and a safe environment as fundamental human rights, contingent on the availability of resources on the part of the state. The state has a duty to take reasonable legislative and other measures to achieve the progressive realization of the right. Under the South African water law, a minimum quantity of water is reserved for both human and ecological needs prior to allocation to other uses, and is provided free of charge to municipal water authorities (DWAF 1999). However, the municipalities still charged users for distribution costs, by requiring the use of pre-paid cards to obtain water from meters. This practice led to an outbreak of cholera when those who could not pay obtained water from polluted puddles and canals (Thompson 2003). The government now provides a minimal "lifeline" of free water, equivalent to half of the minimum standard established by the World Health Organization.

Uses of water beyond the amount reserved for human and ecosystem needs must be registered and, with some exceptions for some existing uses, licensing and fees are required for stream flow reduction activities; irrigation, industrial, mining, and energy uses; and water services authorities. Exceptions for existing uses, and tensions with users not exempted, suggest that whether licensing can truly serve as a reallocation mechanism may depend less on formal changes in the law than on the strength of public participation. This implies the need to provide users with information about the distribution of costs and benefits and to build institutional capacity so as to strengthen the bargaining power and negotiation capacity of the poor (Schreiner and van Koppen 2000).

private ownership have also failed to create incentives for control of off-site impacts of land use practices. With the absence of the recognition of common as well as private and public forms of property rights, there is likely to be insufficient incentive for practices needed to insure continued provision of freshwater services.

Given that property rights are intended to provide security, and cannot arbitrarily be taken away, they do not change easily. Conversely, attempts to recognize or redefine rights will generally be a source of conflict. Such changes are often linked to political momentum generated by extreme events such as droughts, floods, chemical spills, and broader social and political changes, which create opportunities to open up policy debates, and to redefine rights and responsibilities.

Effective property rights institutions with clear and transparent rules can increase stakeholder confidence that they will have access to the benefits of freshwater services and, therefore, willingness to pay for them. Their development can be facilitated by an analysis and recognition of existing rights, both formal and informal, which is indicated by the assets to which various stakeholders have access. This will be more evident in the response to extreme conditions. The development of a feasible strategy for protecting freshwater services will also require an identification of anticipated transaction costs.

Special attention should be given to broad social changes and structural reforms, as these may provide special opportunities for changes in property regimes that are no longer relevant in the face of rapid global changes and the new kinds of problems that are associated with environmental degradation. Post-armed-conflict situations may present special opportunities for changes in policies and rights because such situations reflect a complete breakdown of legal and political systems that were inadequate to begin with. Increased dependence on natural resources in the face of such collapse can also heighten awareness of the role of freshwater and other ecosystem services in human well-being. As was suggested by Professor Kader Asmal, who chaired the World Commission on Dams (2000), in spite of conflicts that have been inherent in the management of increasingly scarce water resources, and the rhetoric of "water wars," the negotiation of water agreements have ultimately been a catalyst for peace and cooperation (Asmal 2000). One reason is that negotiations regarding management of water are rooted in specific water projects, found in a local and site-specific context, which gives them more weight than vague and undefined agreements that have broad scope, and provide a point of departure for a bottom-up decision-making process.

In the face of conflicting claims, allocation of water and development of water-related infrastructure tends to give disproportionate influence to those with the more tangible and dominant economic interests compared with livelihood and environmental concerns. Such differences in power are often difficult to overcome in the short term, even through collective action. Therefore, the process of defining rights to the benefits of freshwater services, and responsibilities for actions needed to insure continued provision, should be regarded as long-term and on-going.

7.2.3 Increasing the Effectiveness of Public Participation in Decision-making

Stakeholder participation may improve the quality of decisions because it allows for a better understanding of impacts and vulnerability, the distribution of costs and benefits associated with trade-offs, and the identification of a broader range of response options that are available in a specific context. If poverty alleviation and sustaining livelihoods are the primary objectives, it is also essential that stakeholder concerns be a starting point for determining what the specific objectives are, and in the development of responses to freshwater degradation.

An important distinction in participatory processes is between those who allow for effective participation and those who only give an appearance of participation in decisions that have, in effect, already been made. Ultimately, the effectiveness of stakeholder participation depends on whether it makes any difference in decision-making, whether it contributes to the establishment as well as the achievement of objectives, and whether it provides an opportunity to work through difficult issues rather than avoid them. It should also provide stakeholders with an opportunity to learn, and to reconsider the values they place on freshwater services. As the Great Lakes water quality agreement illustrates (see Box 7.7), whether or not stakeholders are able to have an active role in the process can also have implications for whether or not goals are achieved (Sproule-Jones 1999).

7.2.3.1 Effectiveness

Principle 10 of Agenda 21 defines participation to include access to information, participation, and justice. It may be limited by factors such as: geographic isolation, common in upper watershed areas; language and educational barriers; access to information that is timely and relevant; whether participation is made possible in the early phases of a process (planning and defining problems); whether the decision process provides an opportunity for deliberation and learning; and legal frameworks that define rights (land tenure, for example) and provide measures of recourse, all of which determine the relative bargaining power of various stakeholders.

BOX 7.7

Implementation of the Great Lakes Water Quality Agreement

The Great Lakes Water Quality Agreement was implemented by developing separate remedial action plans for each of 43 areas of concern. A comparison of the process of developing the plans in each area suggests a strong relationship between success in the restoration effort and the active involvement of stakeholders both in development and in oversight of implementation. Conversely, failure was associated with agency indifference or hostility toward the participatory process. In most cases, agencies regarded the planning process as a source of new resources and public support for traditional concerns, rather than as an incentive for new approaches. In some cases, stakeholder forums were held after the agencies had written the initial reports. In one exception, Hamilton Harbour, more active participation by stakeholders in the development of the plan, and attempts to make the document legally binding, became a source of conflict with the Ontario government (Sproule-Jones 1999).

Conflict played a positive role in improving the plan because it provided an opportunity to work through the more difficult issues and avoid lasting polarization. A low level of conflict in another area, the Saginaw Bay, is attributed to the limited focus of the steering committee, which limited its task to data review and avoidance of difficult issues. Stakeholder input was also done in a way that limited interaction among stakeholders and therefore the opportunity for an iterative learning process (MacKenzie 1996).

In sum, it was the decentralized nature of the overall program that allowed for different strategies to be pursued within the common domain of the planning framework, and for many lessons to be learned. The process of prioritizing plan recommendations also demonstrated their role in the overall basin-wide strategy, and led to greater interaction among governmental agencies.

The nature and quality of participation, and who participates, will be closely related to the scale and institutional context in which it takes place. This may range from informal community-based initiatives to more formal watershed councils and interorganizational partnerships typically found at smaller scales, to river basin organizations found at larger scales, which often cross national boundaries. Smaller scales provide opportunities for more direct participation and face-to-face interactions among those who share relationships to a specific place, and who often have greater understanding rooted in local knowledge. Larger scales present the greater challenges of achieving adequate representation of sub-basin interests as well as of providing information. Both present challenges of accountability and of providing adequate measures of recourse.

A key enabling condition, without which participation is unlikely to be sustained, is effective decentralization of authority for management of freshwater services. This implies the transfer of authority in the form of constitutionally protected rights, downward accountability to those who are represented, and access to the benefits of natural resources (Ribot 2002). This can increase the capacity of local authorities to respond to the variability of site-specific conditions because it enables them to make decisions regarding the allocation of resources, which can also be a source of the financial autonomy that is needed to exercise authority (Kaimowitz and Ribot 2002).

One barrier to democratic decentralization is a widespread perception that local authorities do not have the capacity to manage freshwater resources. However, in the absence of the authority to actually do so, and access to the benefits of natural resources, they have not had the opportunity or the incentive to either gain experience or to demonstrate capacity (Ribot 2002). Learning will therefore be an important part of the process. In many cases, however, farmers and users have built and have been managing their own water sources for centuries, displaying considerable technical skill. The trick is getting this recognized by government agencies, and linking these local understandings to basin-scale management. Access to benefits is discussed more specifically in subsequent sections that pertain to financing and the development of economic incentives.

Broader participation in public policy decisions is mandated in a number of relatively recent international water agreements and in legislative initiatives. Recent years have also seen a proliferation of various kinds of river basin organizations, which are discussed further in the next section, as a more specific institutional form for participation. Among the more far-reaching initiatives is the EU Water Framework Directive, which requires public participation in the development of management plans for all river basin districts in EU countries. This trend toward a shift in water management policies, from command-and-control regulations under the authority of states, toward more integrated approaches that transcend their boundaries, is also leading to the creation of new institutional entities or rules of the game that engage different sets of actors.

This shift raises a number of unanswered questions as to the degree of representation reflected by those who participate, to whom they are accountable, and, ultimately, of the legitimacy of the process (Kaika and Page 2002). Governments at national levels tend to have geopolitically driven interests that often conflict with the well-being and livelihood interests of many of their citizens, to whom they are not always accountable. Local governments are not necessarily any more representative of or more accountable to local stakeholders than are more centralized ones. Independent voices such as those of NGOs may play an important role in supporting community-based efforts and in developing innovative policy approaches, but usually represent narrower interests and are not designed or intended to represent all stakeholders or to create substitutes for democratic processes. Given the scope of the topic, it is not possible to draw general conclusions. However, there are some lessons that can be drawn from selected case studies.

Although decentralization is an inherent goal of many basin-level initiatives, these are often implemented such that pre-existing top-down centralized structures remain in place. For example, in a case study of the state of Madhya Pradesh in India, plans supposedly developed by local watershed committees established for purposes of local participation tend to follow blueprints created by upwardly accountable technicians. These blueprints were driven by the need to meet tangible physical targets and deadlines that had already been decided upon (such as the number of trees planted, number of compost pits dug, hours of volunteer labor) and to obtain the approval of technical committees. Selection of sites for implementation of watershed conservation projects tended to be based on administrative expedience. As a result, areas of tenure dispute on the steeper slopes, where conservation efforts were most needed, tended to be avoided altogether (Baviskar 2002).

Recognition of the limitations of managing irrigation by way of centralized government, and the inadequacy of building irrigation infrastructure without corresponding local management capacity, has led to a trend toward institutional reform in irrigation, in which responsibilities for operations and maintenance of irrigation systems are devolved to user associations (Vermillion 1999). Although, in many cases, these transfers have been incomplete, or responsibilities have been transferred without corresponding rights, they provide a considerable base of experience and a basis for identifying elements that are critical for the success of such initiatives. In general, comparative case studies of the more comprehensive devolution programs have shown a reduction of costs to government, an increase of costs to farmers, an increase in fees collected, increased cost recovery, and variable impacts on productivity. However, some benefits were observed even in less comprehensive devolution programs. For example, in one case, acquiring water through a user association resulted in the reduction of costs to users even though fees were doubled because it eliminated other costs associated with numerous informal transactions, such as the need to pay bribes in return for receiving water allocations (Vermillion 1999).

7.2.3.2 Findings and Conclusions

Degradation of freshwater and other ecosystem services generally have a disproportionate impact on those who are, in various ways, excluded from participation in the decision-making process. Effective participation may therefore require concerted political pressure to overcome resistance by those in positions of power and authority, which may be brought about through various forms of collective action by stakeholders.

Whether or not initiatives to increase broad-based participation have a lasting impact will depend on the extent to which participation is institutionalized in democratic local government as a constitutional right (Ribot 2004). As discussed in the earlier section on rights and responsibilities, in the absence of momentum that may be generated by periods of crisis and broader social change, or other special opportunities for reform, this is generally a process of institutional development that may take time. However, even partial transfers of authority can have benefits. A focus on implementation provides an opportunity for adjustment as lessons are learned.

In the meantime, a key focus for improving participatory processes is to help level the playing field through measures to increase the transparency of information; improve the representation of marginalized stakeholders; engage them upfront in the establishment of policy objectives and priorities for the allocation of freshwater services, with which specific projects should be consistent; and create a space for deliberation and learning that accommodates multiple perspectives.

7.2.4 River Basin Organizations

Recognition that there are trade-offs among multiple interests and uses of fresh water provided by river basins, has led to a trend toward the formation of river basin organizations as a vehicle for basin-wide assessment, planning, management, and conflict resolution. RBOs may be formed to manage basins within individual countries as well as in an international transboundary context, which presents an added layer of management challenges and gives rise to sovereignty considerations.

Pivotal concerns for basin-wide organizations are typically issues of water allocation among multiple and often conflicting uses. Decisions about infrastructure development such as dams, reservoirs, and navigation canals are critical because these modify and divert flows of water that make possible agricultural irrigation and urban development, and are therefore a major driver of land use change. They also block flows of sediment that maintain downstream river channels and coastal areas. Other key concerns at the basin level are with cumulative impacts of land use changes and inputs of pollutants that are detectable over long distances.

A river basin focus was formally recommended at the International Conference on Water and the Environment held in Dublin in 1992, and also endorsed in the 1992 Rio Declaration and Agenda 21 (particularly Chapter 18). However, such organizations are not entirely new. Among the better known early examples are the Tennessee Valley Authority, a regional river basin authority formed in 1933, and the French system of water management, organized around river basins, which was established in 1964. The Mekong Committee, predecessor of the Mekong River Commission, was created in 1957, one of the earliest formal RBOs in the developing world (Ratner 2003). The TVA served as a model for several RBOs in other parts of the world, including regional development corporations in Latin America, river basin commissions in Mexico established toward the end of the 1940s and the beginning of the 1950s (Barkin and King 1986; Garcia 1999), and the Damodar Valley Authority in India. Globally, the International Network of River Basin Organizations lists 133 members in 50 countries at present. The trend is driven in part by donors such as the Inter-American Development Bank, which has already financed, or is likely to finance, more than 20 projects at the basin management level in Latin America and the Caribbean.

RBOs range in type from those that have the authority to plan, promote, and enforce their plans, to those that lack authority but play important roles in an advisory capacity in the assessment needed to inform planning, priority setting, stakeholder negotiations, and decision-making. Examples of the former include the TVA (see Box 7.8), the Murray-Darling Basin Commission, and the French system of water management, which is carried out by river basin committees, and, at sub-basin tributary levels, by local water committees, consistent with national water policies. The recently adopted EU Water Framework Directive also requires a comprehensive basin approach in which the territories of all European countries are assigned to river basin districts that have the authority for implementing the directive. Other countries that currently provide legislative mandates for a nation-wide basin approach are South Africa, Brazil, and Mexico.

In an alternate model, which can be considered an adaptive approach, many RBOs have evolved from a narrow and sectoral

BOX 7.8

The Tennessee Valley Authority

The Tennessee Valley Authority was formed during the Great Depression, and became a model of comprehensive river basin development to meet multiple objectives in support of economic development. The TVA was granted broad powers under which it developed a system of multipurpose dams for purposes of flood control, power generation, navigation, recreation, and maintenance of flows necessary for maintenance of water quality and aquatic habitat. An equally important mission in the earlier period was a program for multi-resource conservation and development, to protect the natural resource base that was recognized as essential for regional economic development (as, for example, by planting trees on eroded lands) and generally to promote human welfare in a poverty-stricken region. However, in the post-World War II period, power generation became the dominant mission, as it was largely self-financing. Although the TVA had had a number of successes and much popular support for its natural resource programs, these have been more vulnerable because they depend on Congressional appropriations for their budget (Miller and Reidinger 1998).

focus on water bodies and point sources of pollution, to a focus on entire watersheds based on inter-sectoral approaches, to the extent that interested parties are able to reach agreement and cooperate. Capacity building thus tends to occur as needed, in response to recognition of new types of problems that are beyond existing response capacities, or in response to extreme events that expose structural weaknesses and bring problems to broad public attention more rapidly, thereby enabling the development of new agreements and policies needed to resolve conflicts among multiple and diverse uses. Regulatory authority tends to rest with existing agencies and political jurisdictions, which may need to be politically compelled to effectively participate, or offered some form of incentive.

This kind of RBO tends to play an intermediary role by activities such as convening interested parties for purposes of strategic planning, management, and conflict resolution, preparing master plans, reviewing project proposals, proposing policies, creating an information system, and coordinating monitoring efforts. One well-known example of this is the Chesapeake Bay Program. (See Box 7.9.) A basin-wide entity that evolved in a similar fashion after the failure of attempts to create a basin-wide planning agency is the Laguna Lake Development Authority in the Philippines, which was given expanded regulatory powers step by step, as its focus shifted from fisheries promotion to watershed-level pollution control (FAO 2002).

Given the absence of an overarching legal authority and the need to rely on voluntary agreements among countries that occupy a basin, this alternate model also tends to be found in transboundary initiatives. A well-known example is the International Commission for the Protection of the Rhine, which was established in 1950 to address concerns about pollution. Over time, however, and in response to specific events, the ICPR began to also address ecosystem concerns, and to develop an integrated approach to river management that considers land use and spatial planning so as to begin to restore floodplains and wetlands, and thereby mitigate flooding by ''making space for the river'' (Wieriks and Schulte-Wülwer-Leidig 1997). In what may be the beginning of a more proactive approach, studies are also investigating the management implications of climate change, which is expected to bring about significant changes in the availability of water (Middelkoop et al. 2000).

A challenge that is particularly pronounced in transboundary water management, to which this section offers special attention, is the strengthening of provisions for various aspects of public involvement, which includes access to information, public participation, and access to justice or legal recourse. An important tool for public involvement is the development of a process for transboundary environmental impact assessment (Cassar and Bruch in press).

BOX 7.9

The Chesapeake Bay Program (Hennessey 1994)

The Chesapeake Bay Program has origins in an agreement made between two key states (Maryland and Virginia) in response to conflicts between oyster fisheries and discharges from urban areas. This was the beginning of a trend toward more inclusive inter-jurisdictional agreements that now cover most of the basin. Given the resistance of existing authorities to the creation of a new regional authority, the program adopted a multistate and federal cooperative governance structure. As concerns extended from the main water body to reduction of nutrient inputs from the upper basin areas, the agreement was expanded to include local governments, who retain responsibility for land use decisions and whose cooperation is needed to meet nutrient reduction targets.

7.2.4.1 Effectiveness

RBOs are constrained or enabled primarily by the extent to which all of the relevant stakeholders participate, are able to agree on objectives and management plans, and cooperate in their implementation. In a transboundary situation, the capacity to implement agreements and plans will depend on the level of commitment of individual countries to integrated river basin management and to ecosystem management, whether they have mutual or complementary interests, and relative bargaining power. Top-down governmental and institutional mechanisms need to be balanced with bottom-up considerations of transparency, public participation, and accountability. It also helps to have specific and measurable goals, such as the cap on water diversion in the Murray Darling Basin in Australia.

An illustrative case is the Mekong River Commission, whose member countries do not include China, which holds 22% of the basin area and is actively building dams. Using its superior bargaining power, it has been negotiating agreements to improve navigation with neighboring states independently, and has avoided joint review of the impacts of its dam-building program on the livelihoods of those in downstream areas, particularly in Viet Nam and Cambodia, which rely on the flooding patterns that sustain fisheries and production of rice, and also control seawater intrusion. China's perspective is that the regulation of the upper Mekong will benefit people by supporting navigation and irrigation activities, flood control, and dry season power generation, and also by containing erosion. Downstream countries have not been given the opportunity for an independent assessment. However, it is important to keep in mind that Thailand and Viet Nam have also been constructing dams and have found ways to avoid the joint review of water diversions that have already created problems downstream. Cambodia, which is the most dependent on freshwater services to support livelihoods, is also the poorest and has the least bargaining power (Ratner 2003).

In the case of the Danube, where the basin states have greater mutual interests and incentive to participate, transboundary management practice has even contributed to political stability—as demonstrated by efforts to establish a river basin agreement that were sustained even during the period of war between 1991 and 1995 (Murphy 1997). Implementation of that agreement, a jointly developed strategic action plan, and efforts to achieve consistency with the EU's Water Framework Directive are also playing an important role in the economic development and strengthening of democratic institutions in Eastern Europe (World Wildlife Fund 2002).

A key enabling condition is access to information and cooperation in the assessment process itself. An important planning tool, developed in recent years, is the transboundary environmental impact assessment (TEIA), which can be utilized to enhance the cooperation and management of shared waters in a transboundary context. A key difference between the TEIA and the EIA is that the TEIA can facilitate cooperation and dialogue across borders. If utilized effectively, it has the ability to provide local people and under-represented interests an opportunity to be heard and to participate in decision-making that affects their environment and livelihoods across borders where otherwise they would be excluded. TEIA is more challenging to implement than domestic EIA because it increases the need for institutional coordination, sensitivity to sovereignty, public participation across borders, varying domestic EIA standards, and cultural differences between involved parties.

Europe has developed the most authoritative, binding commitments to TEIA and several other countries are in the process of developing agreements. In 1991, the UNECE convened the Convention on Environmental Impact Assessment in a Transboundary Context, also known as the Espoo Convention (UNECE 1991), which is arguably the most authoritative and specific international legal codification of TEIA. The Convention's strength lies in the specific codification on issues such as harmonization of national laws, and on the negotiation of more specific bilateral agreements between states, that leave little doubt as to what is required to implement a TEIA. The overall objective is to enhance cooperation between states over shared water courses, harmonize individual EIA procedures between states, and advance nondiscrimination to insure that all affected people have the opportunity to participate equally (Knox 2002). Optimally, TEIA should accord the same protections and access to information to the public of neighboring states as to individuals within a country's own borders. The future development of TEIA will most likely be driven by example, and unfortunately, examples are scant at present. As more TEIAs are undertaken, experience in implementing these will increase (Cassar and Bruch in press)

A pattern often observed is the tendency for basin-level management to be dominated by the more tangible and economically dominant interests. For example, in the case of the Tennessee Valley Authority, which was intended to meet a broad range of multiple objectives related to both conservation and development, emphasis eventually shifted toward provision of hydropower and flood control. This was criticized for providing disproportionate benefits to populations concentrated in large downstream urban areas (Barrow 1998). A similar pattern is evident in a review of basin-level initiatives in five African countries (Barrow 1998). In Central America, watershed management concerns go back to the early part of the last century, but did not get placed at the top of political agendas until they were seen as threats to higher priority interests downstream—such as the sedimentation of large hydroelectric dams or the Panama Canal (Kaimowitz 2004).

7.2.4.2 Findings and Conclusions

Effectiveness of basin-level organizations will depend on the kinds of development paths made possible by water allocation decisions, acceptability of the resulting distribution of costs and benefits among stakeholders, whether these help to achieve objectives of poverty alleviation, and maintenance or restoration of at least those ecosystem processes that support the provision of desired services.

Given that conflicts often exist between basin-wide interests and those at local scales, at which impacts of management activities are experienced and tend to have direct livelihood implications, effectiveness of RBOs will also depend on whether sub-basin and community-level interests are adequately represented and are able to effectively participate in basin-scale decision-making. Sub-basin level organizations such as watershed councils, land-care groups, village level catchment committees, and associations of users and farmers play important roles in addressing problems associated with land and water relationships that are difficult to detect at larger scales, in the promotion of new land use practices, and in the direct involvement of stakeholders in face-to-face settings. In contrast, basin-scale actors tend to be representatives of interested parties, which may include government agencies, NGOs, and associations of resource users. So far, there is little evidence of successful scaling up from village to basin levels (Swallow et al. 2001).

Given that many problems in river basin management are the result of unanticipated consequences, and may not be obvious because they disproportionately affect marginalized stakeholders with little voice in decision-making, a key to RBO effectiveness will be in whether there is an independent and transparent process of assessment in the implementation phase, the relevance of such assessments to actual stakeholder concerns, and the ability to learn from them and to improve on past practices.

Ultimately, whether or not RBOs achieve the multiple objectives of integrated water resources management will depend on how particular initiatives are implemented in practice, an assessment of which will also require stakeholder insights. Other aspects of effectiveness, discussed in other subsections, include decentralization of decision-making processes and use of appropriate regulatory and financial instruments to create appropriate incentives and provide some measure of financial autonomy to RBOs.

Given environmental heterogeneity; embeddedness in social, economic, and political frameworks; different stages of socioeconomic development; and different management capacities and technical expertise, no single institutional model is likely to be equally applicable in all basins. RBOs are actually a mosaic of overlapping institutions.

Regional authorities, based on blueprints, tend to have many of the same weaknesses they were intended to address, such as centralized authority, domination by special interests, and application of sectoral approaches to multiple sectors. They may also be a response to trends among donor organizations toward financing basin-level projects rather than the most pressing problems. The more successful approaches have no blueprints, but tend to evolve in response to site-specific conditions, trends, and extreme events. A key factor, addressed in a previous section, is the democratic decentralization of authority, as it can enable responses more appropriate to their context.

An important enabling condition is that river basin organizations are inclusive of sub-basin level interests. This can at least potentially be achieved by supporting and reinforcing successful community-level initiatives—allowing them to "scale up," and further building a community-level response capacity through provision of information that enables them to effectively participate in responding to larger-scale threats.

Access to information is an important aspect of integrated river basin management that has been increasingly incorporated into water resources management policy and regulation, and is a mainstay of many IRBM policies. Likewise, public participation is being increasingly incorporated into IRBM policies. Access to justice is less commonly incorporated. Many institutions, including the Nile River Basin Initiative and the Mekong River Commission, have embraced principles of transparency and public participation in environmental decision-making. However, while these principles are increasingly common in institutional mandates governing transboundary waters, specific measures to achieve these goals often require more elaboration for implementation to be fully effective. Strengthening the institutional mechanisms to do this is increasingly recognized as the way forward to improve the management of shared water in a transboundary context. One of the ways this is being done is through the development of a process for transboundary environmental impact assessment.

7.2.5 Regulatory Responses

Command-and-control regulatory responses applied to freshwater services include technological, end-of-pipe controls and discharge permits that have been applied to point sources of pollution, regulation of non-point sources through instruments such as the establishment of total maximum daily loads under the U.S. Clean Water Act, and restrictions on land use for purposes of watershed protection. (Non-point sources are more thoroughly discussed in Chapter 6 of this volume.) Regulations also play important roles in creating cap-and-trade systems that serve to limit pollution or resource uses (discussed below). Regulatory approaches also generally support market-based approaches and other instruments by defining the "rules of the game."

7.2.5.1 *Effectiveness*

Regulatory approaches have generally been considered effective for reducing pollution loads from the more significant point sources. However, by itself, this approach is generally regarded as inadequate for addressing numerous small-point sources and non-point sources because it would require more extensive enforcement capacity, as well as site-specific information and authority for controlling land use (NAPA 2000). This is a problem even in countries with well-developed regulatory infrastructure. However, it is perhaps most dramatically illustrated in the Danube Basin, where the privatization in Eastern European countries that followed the end of the Cold War increased the number of economic actors and pollution sources, while the capacity for inspection and enforcement remained low, given the lack of local-level institutions (Koulov 1997).

Key limitations on regulatory instruments with respect to diffuse non-point sources are the lack of flexibility, the information base, and the capacity needed to address the site-specific nature of watershed problems. Regulatory bodies also often lack specific kinds of legal authority as well as the capacity needed to control the diverse kinds of land use activities associated with non-point sources, as this tends to be an authority exercised through local government planning processes, and in which individual landowners retain significant levels of discretion. For example, the U.S. Clean Water Act requires states to establish total maximum daily loads as a basis for allocating the burden of reductions among non-point source emitters. Given that federal agencies have no direct authority for regulating uses of land that result in non-point source emissions, or for water allocation, this provision may be the only source of authority through which such reductions can be legally compelled, and may be useful as an incentive for emission reductions. However, it is also regarded as unwieldy from a technical and administrative perspective, and as having the potential to paralyze efforts by citizens groups with paperwork (NAPA 2000).

TMDLs have been criticized both for the lack of criteria for determining whether objectives have been achieved, and for the lack of independent assessment. A key limitation is that uncertainty in watershed models used to link pollutant loads with water quality so as to demonstrate the effectiveness of required actions for meeting standards makes TMDLs vulnerable to court challenges. It is also difficult to determine whether water quality standards themselves have been achieved, given the variation in standards across states; the lack of a consistent, nationwide set of data on water quality; and the lack of consistent protocols for gathering such data (NAPA 2000). However, it should be kept in mind that, given the heterogeneity of environmental conditions, sources of pollution, and end uses, no single standard approach would be possible or desirable.

Establishment of protected areas in upper watersheds is also a form of regulatory control over land use, as is illustrated in the case of the Hindu Kush Himalaya region. (See Box 7.10.) Such regulatory abrogation of formal or informal property rights is generally regarded as ineffective because it fails to recognize the rights of local populations who have depended on such areas to support their livelihoods, and excludes them from access to benefits (as discussed above).

7.2.5.2 *Findings and Conclusions*

Regulatory approaches that involve market-based incentives such as damages for exceeding pollution standards are particularly suited to the point discharge of pollutants into water bodies. Regulatory approaches that simply outlaw particular types of behavior can be more unwieldy and ultimately burdensome, as they may fail to provide an incentive for finding more effective ways of achieving protection of freshwater services, as a way to avoid them. For example, if not paralyzed by procedures and technical requirements, efforts of citizens groups can complement TMDLs, when they are able to foster stakeholder agreements on actions to be taken and on funding priorities. An alternative to absolute regulatory land use controls is to provide some form of compensation to cover the cost of conservation practices, an approach discussed in the next section.

Non-point sources and small, scattered point sources are difficult to respond to adequately under both regulatory and economic approaches to water management because they require extensive monitoring. It is difficult, for example, to assess quantities of nitrates leaking from a given field, as it depends on rainfall, management practices, and other site-specific conditions. Because of the uncertainty with regard to non-point source emissions, regulatory measures may be more effective than economic instruments in controlling them. Where appropriate, they may also become more effective as scientific and technological advances make it cheaper to gather and disseminate information.

7.3 Economic Incentives for Supply and Demand Management

Typically, water as a resource has been undervalued and underpriced, while infrastructure projects for water resources development have been heavily subsidized. This disconnect has led to water being managed ineffectively for people and ecosystem services (Johnson et al. 2001). Economic incentives generally refer to the use of market-based instruments, incentive payments, and pricing strategies to alter the economic return from the use of scarce resources to better reflect the environmental and social impacts.

These strategies are being applied to manage fresh water in at least four different ways:

- using markets to reallocate water from existing, low-value uses to new, higher-value uses, such as from agricultural to urban or instream uses (water transfers and water banking, for example);
- developing cap-and-trade systems to avoid overexploitation of water resources, improve water quality and mitigate for ecosystem degradation (nutrient trading, groundwater mitigation banking, wetland mitigation banking);
- using incentive payments and water pricing to provide water and watershed managers with incentives to conserve water quantity and improve water quality as it is conveyed to the point of use, thereby providing a way to meet additional uses with the same amount of water (such as incentives for agricultural water conservation); and
- developing public/private partnerships for the financing of new supply infrastructure and technologies, particularly for municipal and industrial purposes.

BOX 7.10

Causes of Environmental Degradation in the Hindu Kush–Himalaya Region

A widely accepted theory in the 1970s was that accelerated erosion, sedimentation of river beds, and increasingly severe downstream flooding in the Hindu Kush–Himalaya region was driven by population growth, ineffective agricultural technologies, cultivation of steep slopes, forest clearance, overgrazed pastures, and unsustainable use of forests for fuelwood and fodder. This theory of Himalayan environmental degradation was found to be largely unacceptable by international experts on the grounds that such impacts were not significant when compared to the high rate of mass wasting and natural erosion, which delivers large amounts of sediment to river systems in episodic events (Bandyopadhyay and Gyawali 1994); the experts said there were more complex root causes (Jodha 1995; Kasperson 1995).

Nevertheless, the concept that accelerated erosion was largely due to unsustainable practices and population growth retain significant influence on policy-making in Asia. For example, in the mountains of India and China, the concept provides justification for government efforts to increase access and control over watershed forests, and to manage them in a way that supports national-level interests such as revenue generation and prevents sedimentation of dams. In India, watershed management is for the stated purpose of preventing accelerated erosion, flooding, and desiccation of water supplies, and has been carried out through conservation policies that had excluded local populations from access to benefits that support livelihoods, although there is now greater emphasis on more participatory modes of forest management (Vira 1999). In China, logging bans and grassland enclosures were adopted in response to large floods, and to prevent siltation of the Three Gorges Dam, as well as to restrict indigenous land use practices of shifting cultivation and nomadic pastoralism, which are seen as the culprits (Blaikie and Muldavin 2004).

Given the differences of scale, it is not at all clear that changes in upstream management practices would detectably or significantly reduce sedimentation of dams or distant downstream flooding, though it may significantly reduce more localized flooding. Although it is difficult to generalize the reasons for the extent to which land use practices and deforestation contribute to the flooding in the foothills, there is not much doubt regarding the absence of convincing ecological links of such upland degradation with the regular monsoon floods in the distant deltaic plains, such as in Bangladesh. According to a recent case study, neither the frequency nor the volume of flooding has increased in Bangladesh over the last 120 years. The study also found that following a period of heavy rainfall and catastrophic flooding along a tributary of the Ganges in Nepal, there was only an insignificant fluctuation of water levels in the Ganges itself, which could have been associated with local rainfall (Hofer 1997).

An alternate explanation for the flooding in Bangladesh is rainfall within Bangladesh itself, and in the Meghalaya Hills in the Brahmaputra Basin, which are located in India, north of Bangladesh, a place known for some of the highest rainfall in the world. It also has shallow soils and rocky surfaces, which leads to immediate runoff (Hofer 1997). Regulatory restrictions on land uses that support local livelihoods also do not address root causes of degradation that may be more significant, such as land use intensification to support a shift to production for markets rather than for local consumption, and conflicts associated with the nationalization of forests, privatization of common property, and development of roads and industries, which are in conflict with livelihood interests (Jodha 1995; Kasperson 1995).

These strategies may be used to manipulate the economic incentives affecting the production, allocation, or consumption of freshwater services. Direct payments for watershed protection change the incentive structure for management of upland areas with resulting changes on downstream availability and quality of water. The existence of functioning water markets places a financial opportunity cost on the holding of water rights and, therefore, makes the allocation of water rights more responsive to the economic values associated with different uses of water. Direct incentives for household or on-farm water conservation cause consumers to adopt water-saving technologies that reduce the overall demand on water delivery systems and their natural sources.

Financing for these strategies may involve a mix of public and private monies. When the benefits can be limited to those who pay for them, consumers of freshwater services or entities that generate power or distribute water often pay directly for these services. Acquiring ownership rights to use water or manage lands for watershed services through land and water markets, paying others to conserve water or improve land management, or acquiring mitigation credits through a cap and trade system (effectively trading one service for another) are examples of permanent and temporary ways to acquire access to freshwater services. Public financing can be used to provide freshwater services through these pathways, but often, its primary route is through the more direct route of funding water resource development that increases water supply to consumers or water and watershed conservation activities that improve water supply and quality. Public "funding" may also be obtained for actions to restore freshwater services through tax incentives to companies and individuals. Clearly, another important public role is to provide the enabling environment (property rights and regulations) for markets and cap and trade systems.

Nongovernmental entities and other private/public intermediaries also often play a role in gathering funds from service consumers, public sources, and philanthropists, and then disbursing funds to farmers and households. These intermediaries can help to reduce transaction costs inherent in establishing arrangements among numerous stakeholders spread out over large and remote areas, who may lack clear title to land or access to water by facilitating agreements among them, negotiating on their behalf and providing various kinds of legal and technical assistance, and assessing and disseminating appropriate information. Financing arrangements in practice often consist of a combination of sources and may finance not just freshwater services, but also a number of other ecosystem services.

As the scarcity of fresh water has increased relative to demand, attention has intensified across all the approaches and methods listed above. Considerable innovation has occurred in the last decade in the area of cap and trade systems (in water and other environmental services). Although these tools hold promise, it is still too early to assess their effectiveness. Box 7.11 reviews a number of the innovative ways that these created markets can affect resource use and environmental degradation or restoration. As cap and trade systems lead to the creation of markets, a proxy for their effectiveness is the extent to which water markets—which have a long history—have functioned in the reallocation of water rights. A thorough examination of water markets is, therefore, provided below.

BOX 7.11

Cap-and-Trade Systems for Water and Watersheds

Cap-and-trade systems have been applied effectively in controlling point source air pollution, but are relatively new as a tool for provisioning water and watershed services. Where ecosystem maintenance or restoration goals are well-defined, the employment of cap-and-trade systems may be an appropriate response. The cap-and-trade approach involves three steps: (1) determination of the cap, or the level of resource use or pollution that is allowable, (2) the allocation of use permits or pollution credits, and (3) the development of a market for the exchange of permits or credits between willing buyers and sellers. Key issues in designing these systems include the initial method for allocating rights and rules for transferability. Limits on transferability may be used to prevent concentration of rights in the hands of a few or to maintain rights within a particular community. However, such limits may also reduce efficiency by reducing the pool of buyers and sellers (Rose 2002). Cap-and-trade systems also require a strong regulatory infrastructure to insure that the caps are met.

These systems are being applied increasingly to the management of groundwater, surface water, wetlands, and water quality.

Groundwater Credit Trading: In basins where streams are fed largely by groundwater, once surface waters are fully allocated, additional groundwater withdrawals can have adverse affects on stream flow. Once a limit is placed on total groundwater withdrawals, groundwater pumping credits are created and traded so as not to further impair surface water flows. Such a system is in use in the Edwards Aquifer of Texas, where it has led to an active market in credits (Howe 2002).

Groundwater Mitigation, Credit Trading, and Banking: Another approach is to use a cap and trade system to achieve conjunctive management, which is the integrated management of surface and groundwater. The further development of groundwater sources can then be off-set, not just by reducing other groundwater withdrawals, but also by restoring stream flow or recharging aquifers. In 2002, the state of Oregon developed a mitigation cap-and-trade system for the Deschutes Basin, which has led to the development of markets for both temporary and permanent credits (see www.wrd.state.or.us).

Wetland Mitigation Banking: Wetland mitigation banking was developed in the United States by the U.S. Environmental Protection Agency in order to provide a more cost-effective and efficient option to meet regulations under Section 404 of the Clean Water Act. Mitigation banks may be established where a wetland has been "restored, created, enhanced, or (in exceptional circumstances) preserved" and the credits created can later be applied to areas in which wetlands are removed by development (USEPA 2003). Advantages include eliminating the temporal gap between when a wetland is created and one is eliminated by development, reducing costs of regulatory compliance, and reducing delays for development. As documented later in this chapter, criticism exists in the form of uncertainty about engineers' and biologists' expertise to recreate the intricate functions of a wetland and that the new wetlands may not be anywhere near the original wetlands. For example, in 2002, the New Jersey Department of Environmental Protection released a study that concluded its mitigation banking program, which had targeted two acres of restoration to every one acre lost, had actually resulted in a 22% net loss of wetland acreage and created only 45% of expected acreage (NJDEP 2002). While most wetland mitigation banks are federally supported, there is growing entrepreneurial interest and the first private bank was chartered in December 2002. The major markets that have been identified as targets for wetland mitigation banking include commercial land developers, airports, departments of transportation, oil and gas transmission line companies, and electric utilities (Zinn 1997). According to the U.S. Environmental Protection Agency, roughly 100 mitigation banks are in operation in the United States.

Nutrient Trading: Water quality trading is another response emerging in the United States to meet total maximum daily load regulations under the Clean Water Act. Under the Act, waterways must not exceed certain nutrient levels and states must develop plans for remediating the waterways back to established TMDL levels. Trading is limited to the immediate watershed in which the TMDLs are specified, though there are some exceptions. The Connecticut Nitrogen Exchange Program is one example of a nutrient trading program that emerged to reduce hypoxia in the Long Island Sound. In 1990, Connecticut, New York, and the EPA agreed upon a Comprehensive Conservation and Management Plan to reduce the level of dissolved oxygen in the Sound by 58.5% between 2000 and 2015 (USEPA 2003). To meet its commitment, Connecticut chose to implement a trading system among point and non-point sources, requiring 79 municipal, publicly owned treatment works to reduce nitrogen discharge by 64% from 2000 levels. The exchange is expected to save $200 million over 14 years; in its first year of operation, the program reduced nitrogen discharges by 15,000 pounds, or 50% of the target reduction (Rell 2003; Johnson 2003).

Nutrient trading has also developed in effluent, stormwater, and agricultural runoff. For example, in Australia's Murray-Darling Basin, efforts are under way to develop markets for salinity trading. The Basin's Salinity Debits and Credits Management Framework allows states that have contributed to the cost of projects to reduce salinity, thereby creating salinity credits, to implement measures that might increase salinity within agreed limits, employing salinity debits. Credits may be traded and are tracked through a Salinity Register (Murray-Darling Basin Commission 2001).

An important aspect of freshwater services provisioning is the conservation of existing supplies. There are substantial savings to be gained from improvements in municipal, industrial, and agricultural systems around the world (Gleick 2000). For example, in California, the potential for water conservation and efficiency improvements in just the residential, commercial, and industrial sectors is 33% (Gleick 2003). However, as the single largest use of water in the world, irrigated agriculture has been estimated to be only about 40% efficient on average and, therefore, a prime target for conservation measures (Postel 1997). The remaining 60%, lost through leaky or unlined canals, overwatering of crops, and inefficient technology, is often considered wasted (Molden and de Fraiture 2004). Gleick notes that as of 2000, only 1% of the world's irrigated land was under micro-irrigation; 95% had efficient drip irrigation or micro-sprinklers (Gleick 2003).

While significant savings exist through irrigation efficiency improvements, many studies suggest there may not be as much as often thought (Molden and de Fraiture 2004, p. 9). For example, the lost water typically returns to a waterway or recharges groundwater and is subsequently used downstream, either by other irrigators or ecological uses instream. Therefore, water productivity at a basin-wide level is likely much higher than when estimated at the irrigated agriculture level (Gleick 2000). In fact, improvements in irrigation efficiency may even harm downstream users as savings are used upstream of where they used to be available (Molden and de Fraiture 2004). While improvements in irrigation efficiency will be necessary to improve water productivity, they will not be sufficient to meet environmental, municipal, or other needs since savings, particularly in water-tight areas, will likely provide incentive for irrigators to increase intensity or production. Therefore, regulations or other means will be needed to ensure that savings are allocated to other uses as well as agriculture (Molden and de Fraiture 2004).

An international research program known as the Comprehensive Assessment of Water Management in Agriculture is currently under way, to be completed in 2006, and will provide a thorough assessment of a number of topics including that of conservation and management of water in irrigation. Pending the results of that Assessment, this section on payment approaches takes up an area of increasing innovation internationally, that of making payments to landowners for the watershed (hydrological) services that well-managed lands can provide to downstream users and communities.

The focus of this economic incentives section shifts, then, from rural land, water, and agricultural water management issues to economic incentive issues surrounding the need to provide water supplies to communities for domestic and industrial purposes. Even with greatly improved management of ecosystems upstream, there will still be an enormous unmet and increasing need in coming decades to provide the physical infrastructure to bring water to communities. A critical issue here is the financing that will be required to implement these upgrades and new systems and what will be the roles of the public and private sectors in carrying out this task.

A general recognition that regulatory approaches are, by themselves, inadequate for ensuring the continued delivery of fresh water has led to the recent interest in applying these market-based institutional arrangements. In part, this also reflects societal changes in attitudes and an increasing willingness of beneficiaries to pay for these services (Landell-Mills and Porras 2002). Efforts to develop such arrangements can also be considered part of a global trend of institutional change in water resources management aimed at improving the recovery of costs—both the operational costs of delivering basic water supplies and sanitation, as well as the costs of conservation and research activities (Saleth and Dinar 1999).

7.3.1 Water Markets

Bjornlund and McKay offer three compelling reasons to consider the use of markets for the provision of fresh water (Bjornlund and McKay 2002). First, tradable water rights create a value for water that is distinct from land and, therefore, able to be preserved in its own right. Second, full cost recovery pricing incorporates externalities associated with inefficient use and encourages inefficient users to leave the market. Third, the use of market forces rather than government intervention to facilitate reallocation reduces transaction costs and delays. While markets clearly respond to efficiency objectives, they have their limitations, particularly with regard to the importance of equitable solutions (Johnson et al. 2001). Water is often viewed as at least partially a public good (Thompson 1997). As a result, purely unfettered markets are not only unlikely to evolve, but probably undesirable. The support for market approaches is typically circumscribed by the requirement that these markets take place within a carefully constructed policy and regulatory framework (Howe 2002).

As countries develop, water transfers, and leasing and trading programs have emerged to address the need to reallocate water from traditional uses (primarily agriculture) to new and growing uses (municipal and environmental). The establishment of markets for the transfer of water depends on a number of enabling conditions that largely have to do with creating private property rights that are transferable. Institutional approaches to enhance transferability include water exchanges, water banks, and instream acquisitions programs. Cap-and-trade systems are more novel, but are rapidly being applied to resource and environmental issues related to water management. A description and context for each of these aspects of water markets is provided below, followed by an assessment of the effectiveness of markets as a reallocation and supply response.

While the attempt here to distinguish between various forms of market-based responses is useful for the purpose of analysis, in practice, one response may borrow from or encompass another, and some operational issues such as price discovery may apply across the board.

7.3.1.1 Water Transfers: Property Rights and Enabling Conditions

Globally, the range of systems that regulate the allocation and manage the use of water is broad, with the primary distinction being the degree to which the user has a private right to the use and ownership of water. In systems where water is owned as a public resource and use occurs only upon the issuance and renewal of a temporary permit, the use of markets to reallocate water is an unlikely response. In these systems, water allocation is achieved through regulatory control, government policy, and administrative process.

On the other end of the spectrum are systems where rights to use or ownership are extended to users and are tradable. The ability to trade in these rights will typically depend on their transferability and validity. In cases where water rights are not clearly defined and allocated, trading will be limited if it occurs at all. For example, in some basins in the western United States, water rights are not fully adjudicated or prior Indian reserve rights exist that have not been adjudicated or settled. Lack of clear property rights limits the transferability of these rights and can increase the risk to those engaging in water rights transfers.

The validity of the intended use on a transfer is another important consideration. Many countries and states manage water under a ''beneficial use'' doctrine, whereby water not beneficially used is lost to the user or right-holder. A notable exception is Chile, whose free-market Water Code no longer contains a requirement for beneficial use. The result of this policy was a prolonged and unsuccessful effort in the mid-1990s by the state governmental water agency to pass a tax on unused water rights (Bauer 2004). However, the more unfortunate result was that the large hydropower companies were able to file for large water rights on some rivers, thereby establishing monopoly rights on those rivers.

Where the beneficial use test applies, a key enabling condition for the use of markets in reallocation is the statutory provision that the ''new'' uses are beneficial. This is particularly the case where markets are used to reallocate surface water from out-of-stream uses to instream uses. For example, since the 1980s, the Pacific Northwest states of Oregon, Idaho, Montana, Washington, and California in the United States have adopted laws and regulations that allow water to be transferred instream as a beneficial use, in some cases even adopting instream water rights that reflect minimal needs for fish and wildlife (Landry 1998). A number of U.S. states also recognize recreation, aesthetics, and pollution mitigation as beneficial uses. Prior to these statutory changes, it would not have been possible to transfer the character of use to an instream use.

Ownership of instream rights is a further complicating issue. When water is reallocated to an agricultural or municipal user, the owner is clear. This may not be the case when water is reallocated to an instream use. Despite much interest in the creation of private "trusts" to hold these water rights, western U.S. states allowing instream beneficial uses have preferred to adopt a public trust doctrine, whereby these rights are held exclusively by the relevant state agency upon transfer. The buyer interested in restoring instream flow must therefore purchase the water right and transfer it instream by, in effect, turning it back to the state. Difficulties with this approach exist as conflict may develop between the roles of the state as administrator and as property right holder, and constraints on state budgets may impair efforts to insure that the instream flow rights are monitored and enforced as against out-of-stream rights (Aylward 2003).

Depending on the context, permits and rights may also be transferable on a temporary basis. Where water is leased in this manner, the ownership or use returns to the original user upon termination of the lease. With water that is leased instream, the buyer may become the lessee, but the leased right is often still held by the state for the duration of the lease.

7.3.1.2 Water Exchanges

Water exchanges vary in size and activity from full service operations offering brokerage, water rights information, and consulting services to small, nearly virtual bulletin board systems providing a place for buyers and sellers to connect. Bulletin board systems are pervasive wherever there is an agency (such as an irrigation district or company) that provides centralized services (in particular water delivery) to water users. However, it is in the Murray-Darling Basin of Australia where water exchanges have seen the most rapid pace of institutional development for trades between water users. Two distinguishing features of the exchanges in the Murray-Darling Basin are that they serve to transfer water outside of the traditional confines of a specific administrative or geographic area and they have pioneered the use of electronic auction techniques. These exchanges operate for the purpose of easing the transfer process and facilitating short-term trades that might not take place otherwise. While most trades that take place are temporary—and in some cases nearly instantaneous—the exchanges are beginning to place a few permanent trades (Bjornlund 2002).

Water exchanges emerged in Australia in response to a cap placed on water use within each state of the Murray Darling Basin. The cap was established in 1997 and limited surface water usage to 1994 levels, but left it up to the states to decide how to achieve the reduction. Trading has been active since 1997, particularly in drought years when all allocations are cut back (Bjornlund 2002). While water exchanges do not require a cap-and-trade system per se, it is worth emphasizing that, ultimately, markets evolve only in the presence of scarcity. With respect to water resources, scarcity may evolve in response to a cap on further appropriation or it may evolve due to a physical scarcity of water. In the western United States, water rights are allocated according to the prior appropriation doctrine of "first come, first served." No cap on appropriation of surface water rights is necessary to drive a water market in this case. "New" appropriations will be of little value when stream flow is already fully appropriated—in this case, there will be a natural tendency for new needs to be met by the reallocation of existing prior uses (provided of course that priority is conveyed along with the property right).

7.3.1.3 Water Banks

The term "water bank" has many interpretations but, in general, refers to an institutional arrangement for temporarily moving water from one use (or user) to another that involves the participation of an intermediary. Water banks are a feature of the American West most notably in Idaho, Texas, California, Oregon, and Washington (Clifford et al. 2004). The first formal rental pool in Idaho was set up in 1937, following a decade of informal water leasing between agricultural users (Howe 1997). The term "bank" may well reflect the fact that most large water banks are based on water stored in a reservoir. The water is therefore "banked" or stored until such time as it is purchased and used.

Water banks have become a preferred option in the western United States, as they operate within a confined area—often an irrigation district—and the water is unlikely to travel very far. Since irrigation districts have a vested interest in retaining the right to delivery water and, therefore, ensuring their customer base, they are more likely to find the temporary and limited nature of water banking an amenable option for storage management. The distinguishing feature between water banks and exchanges is that an exchange simply brokers water rights, whereas a bank will either hold rights or retain a role as a lessee of the rights.

7.3.1.4 Instream Water Acquisition Programs

The acquisition of water for instream flow restoration on the part of state agencies and local water trusts and conservancies has become a popular tool in the western United States, where rivers are affected drastically by summer withdrawals for irrigation, which often leads to the listing of species as threatened and endangered under federal law. (See Box 7.12.) Water leasing is particularly popular with water right holders because it avoids the permanent dedication of the water right instream. It may also be useful in extending the lifetime of a water right given that if a water right is not being put to a "beneficial" use, it can be subject to forfeiture (typically after five years). Leases also provide considerable flexibility as they may take many forms, including fixed terms, dry year options, forbearance agreements, conservation off-sets, and exchange or barter agreements. As in general with efforts to "purchase" water for instream flow restoration, leasing is typically found off main stems and may be particularly useful in small tributaries where a small quantity of water may make the difference between a stream that goes dry and a minimum flow level to support fish and recreational uses.

Instream flow acquisition is an attractive alternative in prior appropriation systems because many streams and upper tributaries are over-appropriated. This means that even if a "senior" water user does not choose to use water in a given year, there will be a "junior" user who will divert the water. Only if an "instream" water right is created that is of sufficient seniority to ensure that the state can protect the water from junior users can instream flows be assured. Ultimately, in order to ensure environmental flows sufficient to maintain habitat and species, permanent instream transfers of water rights will be necessary.

7.3.2 Effectiveness of Market Approaches

Water trading has a long but narrow history (Howe 2002). Water auctions were held in Spain as far back as the sixteenth century and water trading has occurred in Chile and the United States for over a century (Howe 2002). In the United States, the Carey Act in the early twentieth century provided incentives for the formation of private companies to appropriate and develop water resources in the American West. Transfer and leasing of water between agricultural users began shortly thereafter.

Reallocation of agricultural rights to other uses—particularly municipal uses—also has a long history, but a contentious one. Water trading in California in particular has an infamous history

BOX 7.12

The Deschutes River Conservancy (http://www.deschutesrc.org)

The Deschutes River Conservancy is a partnership initiated by Environmental Defense, the Confederated Tribes of the Warm Springs Reservation, and irrigation districts in central Oregon in the United States. DRC founders recognized the need for a private organization with ecosystem-determined goals and methods based on positive incentives, consensus, and local governance. Since approximately half of the Basin's land area is managed by federal agencies, it was clear that such a private organization would need the capacity to partner on projects with the federal agencies to be truly ecosystem- and basin-wide in scope. In March 1996, Senator Hatfield introduced S. 1662, authorizing federal agencies to work with this private nonprofit organization on ecological restoration projects using Bureau of Reclamation funds on a 50–50 cost-share basis.

The DRC mission is to restore streamflow and improve water quality in the basin through the use of voluntary market-based economic incentives with the aim of enhancing the quality of the region's natural resources and adding value to its economy. The DRC's Water Acquisitions Program uses instream provisions under Oregon water law to acquire and protect water rights instream. In nearly six years of operation, the DRC has increased local awareness and appreciation for the value of water rights through payments to irrigation districts for water conservation projects and providing a market for temporary and permanent instream transactions for water right holders. These programs have trebled flows in the main stem Deschutes and restored 50% of target flows for fish and wildlife in critical dewatered tributaries. The DRC also operates a full range of brokerage and administrative services, including fee-for-service assistance to private clients, a groundwater mitigation bank, and water banking services to irrigation district and municipal partners to more effectively and equitably manage water resources in the Basin.

DRC efforts to improve water quality are directed through its Riparian Habitat Restoration Program. Since 2002, the DRC has partnered with the Climate Trust of Oregon on an innovative water quality and carbon sequestration program. The Climate Trust is providing the DRC with $780,000 to help landowners in the Deschutes Basin restore riparian areas while reducing greenhouse gases in the atmosphere. The DRC is matching the Climate Trust's funding with federal and foundation funds. Additional support comes from leveraging U.S. Department of Agriculture Conservation Reserve Program funds. In the first year of the program, a large-scale restoration project in the headwaters of the Trout Creek Watershed replanted 100 hectares of contiguous riparian buffer with 71,000 native plants (willow, red osier dogwood, mountain alder, and ponderosa pine, among them). A 52-year conservation easement and monitoring plan now protects 10 kilometers of one of the principal remaining steelhead runs in the Deschutes Basin.

(Reisner 1986). The Owens Valley water ''grab'' by the Los Angeles Metropolitan Water District in 1905 presages the worst fears of those opposed to a free market in water. By quietly purchasing most of the water rights in the Owens Valley, the District severely curtailed agricultural activity in the valley. The action had immediately devastating effects on the local economy and way of life, but even longer-lasting negative impacts on the Mono Lake ecosystem and its migratory waterfowl. The assessment below begins by checking the extent to which water markets have led to significant reallocation and how these transfers have affected third parties.

7.3.2.1 Efficacy of Markets in Reallocation

Several countries have experimented with creating water markets for irrigation water, most notably the western United States, southeastern Australia, Mexico, and Chile (Bjornlund and McKay 2002). With the most laissez-faire system, Chile has relied largely on the natural evolution of markets. Australia and the U.S. states have tried to stimulate markets through the creation of water exchanges, water banks, and instream flow acquisition programs. Evidence of market activity, the balance of trade between permanent and temporary reallocation, pricing, and institutional innovations to facilitate trading are covered below.

7.3.2.1.1 Chile: The free market

In Chile, a formal water policy for markets (declared in 1976) and laissez-faire Water Code (1981) has promoted a free market in water rights largely unfettered by public interest concern (Bauer 2004). Despite the extensive nature of the system, temporary trades between farmers on the same canal are still the most frequent trades that take place, although these are often informal trades that do not depend on the Water Code (Bauer 2004, Bjornlund and McKay 2002).

A study of four areas in central and northern Chile, selected because they were expected to have active water markets, showed that in fact there was very little trading of water rights in three of the four study areas, with the exception being the Limarí area (Hearne 1995). The principal explanation for the lack of activity was that the rigid canal infrastructure made it costly to change water distribution, particularly among farmers. Further work in the Santiago, Chillán, and Bulnes water registries found annual trading to vary from 0.6% to 3% of total allocations (Rosegrant and Gazmuri 1994). In the Santiago registry, which had the highest level of activity, only a small amount of the water (3%) moved from agricultural to municipal use.

A later study of the Limarí water market found that the market operated efficiently and provided important benefits for both buyers and sellers (Hadjigeorgalis 1999). In Limarí, there is abundant evidence that water has been frequently reallocated to higher-value uses within the reservoir system. In addition, the market has provided farmers with the flexibility to manage some of the risks caused by uncertainties in water supplies and in agricultural markets. Poor farmers, for example, have been able to lease their water rights to other farmers during drought years, when water prices are high and income from irrigation is uncertain. Much of the temporary trading occurs informally between farmers on the same canal system and numbers are not available on the size of this market (Brehm and Quiroz 1995). However, Romano and Leporati note that the market has had negative distributive impacts, especially with regard to peasant farmers who have little bargaining power (Romano et al. 2002).

The Chilean experience thus suggests that the freedom to buy and sell water rights has led to the reallocation of water resources to higher-value uses only in certain areas and under certain circumstances (Bauer 2004). Brehm and Quiroz (1995) argue that lack of activity may simply suggest a close to optimal initial allocation of rights. Still, studies have identified a number of factors limiting market activity in Chile, including:

- constraints imposed by physical geography and rigid or inadequate infrastructure (limiting cross-canal transfer);
- legal and administrative uncertainty over the validity of water rights;
- cultural and social reluctance to conceptualizing water as an economic good;
- inconsistent and variable price signals; and

- concurrent subsidies for efficiency improvements (Bauer 2004; Bjornlund and McKay 2002; Brehm and Quiroz 1995).

Despite the lack of market activity, the principal benefit of Chile's Water Code is that secure private property rights have led to significant investment in the water sector for both agricultural and infrastructure development (Bauer 2004; Hearne and Easter 1995; 1997).

7.3.2.1.2 The United States: Water banks and instream flow acquisition

Permanent transfers of water have long occurred in western part of the United States. Allocation and overallocation of stream flows in the late nineteenth century in many areas of the West, combined with the application of the prior appropriation doctrine, enabled a market for transfers. Few historic measures of market activity exist; however. One investigation of transfers in the period 1975–84 in several states suggests widely varying activity. Only three transfers were found in California, where federal and state projects do not allow transfers outside of such projects. In Colorado and New Mexico, where markets are most developed, approximately 1,000 transfers were found, largely from agricultural to non-agricultural users.

Temporary transfers are far more prevalent and often facilitated by water banks. These banks, known as ''officially sanctioned arrangements for short-term transfer,'' were first operated in Idaho in the 1930s (Howe 1997). Over the years, water banking on the Snake River, based on storage, has seen continued development. In the 1980s, much of the water was transferred from agriculture to power generation, with annual traded volumes in the range of 185 to 370 million cubic meters. As the price was set administratively ($2 per cubic meter during the late 1980s) and did not account for water availability, large supplies (with little demand) were obtained in good water years and minimal supplies in dry years (with high demand). This lack of a responsive pricing mechanism has been a major problem with this bank, as well as the Boise and Payette River banks, also operating in Idaho (Howe 1997).

California operated a ''drought'' water bank in 1991–92 and 1994. The principal purpose of the bank was to transfer water from irrigation to municipalities. In 1991, the bank bought over 1 billion cubic meters of water (largely stored water) without having a ready buyer, ending up with 505 million cubic meters of unsold water that year (MacDonnell 1994). Half the water came from fallow land, and the remainder from substituting groundwater for surface water and from storage. The bank paid $0.10 per cubic meter, and sold the water for $0.14 per cubic meter (Howe 1997). In effect, the bank acted as a speculator by holding water across seasons and may have actually competed with its customers. In subsequent years, the price was reduced significantly, with buying occurring at $0.04 and selling at $0.58 per cubic meter.

Despite the obstacles encountered with the drought banks, water marketing and trading has become an institution in California. It accounts for approximately 3% of all water use in the state, with agricultural districts from the Central Valley, Imperial, and Riverside counties supplying 75% of the water. Market drivers have not been municipal users, as is the case elsewhere, but rather changing environmental regulations (Hanak 2003). Trade is balanced between agricultural, environmental, and municipal demands, with direct environmental purchases for instream and habitat restoration (resulting mostly from federal and state programs) growing from 12% to 30% of demand between 1994 and 2001. As much as 50% of demand now comes from agricultural demand growth in the San Joaquin Valley and the remainder from municipal demand, which has actually declined in importance since 1994. However, municipal demand accounts for the longer term and permanent purchases—approximately 20% of the total (Hanak 2003).

There is some consensus on the conditions required for successful water banking efforts. Establishing a spot market (perhaps through an auction) is important in conveying necessary information on trades occurring in known quantities of available water (MacDonnell 1994; Landry 1998). Second, expected future flows should be sold in volumetric terms (something that is rarely done). Third, water rights may be leased, and in doing so, the delivery risk falls on the lessee, something that is commonly practiced in irrigation districts (MacDonnell 1994). Other factors that weigh heavily on the design and operations of a water bank are the homogeneity of the water rights, the ability to hold water over time (in storage, for example), and thinness of the market (MacDonnell 1994, pp. 4–15).

Although water banks in the United States have typically been devised as a means of shifting water to urban and power users, they have also been used to meet ecosystem needs. In 1993, the Bureau of Reclamation made the first purchase from the bank for the purposes of salmon restoration, acquiring 250 million cubic meters for this purpose (MacDonnell 1994). In the 1990s, explicit programs of water acquisitions for instream purposes and ecosystem needs were initiated, and programs to develop the institutional capacity for such efforts are now under way in a number of states, including one multistate program in the Pacific Northwest. (See Box 7.13.) In Australia, the realization that markets have not spurred instream flow restoration led to proposals for the creation of a large state-sponsored fund to acquire water rights for this purpose.

Instream leases are found to be very effective in situations where timeliness, low transaction costs, or temporary intermittent restoration is needed, or when water right holders choose not to exercise their water rights. In addition, leases serve to introduce water right holders, state agencies, and interested participants to the process of instream transfers. Between 1990 and 1997, over 2.5 million cubic meters of water was leased instream in the western states, constituting 84% of total water placed instream, including purchases and donations (Landry 1998). However, leasing constituted only 61% of total expenditures, confirming that leasing is a less costly method of placing water instream.

7.3.2.1.3 Australia: Murray-Darling markets and water exchanges

Permanent and temporary trading was introduced in the state of Victoria in the 1989 Water Act. Following finalization of regulations in 1991, the first transfers began in 1992. In South Australia, a moratorium on new water licenses in 1969 was followed by a period of readjustments to water rights to reflect actual or committed use such that all resources were allocated by 1976. Trading was subsequently introduced in 1983, first outside of large irrigation districts and then in 1989 to these districts. The Irrigation Act of 1994 permitted trade between districts and non-district areas. A cap of 2% of total entitlement limits movement of water out of districts (Bjornlund and McKay 2002).

The Australian experience reveals the predominance of water banking to facilitate temporary trades rather than permanent reallocation. Studies of permanent transfers in the Goulburn-Murray Irrigation District in the state of Victoria and River Murray show annual trades as a percentage of total allocated volumes ranging from 0.35% to 1.7% (Bjornlund and McKay 2002). The annual number of transfers in 1995–96 (the last two years of the study

BOX 7.13

Financing Flow Restoration in the Pacific Northwest of the United States (www.cbwtp.org)

In the Pacific Northwest, the public trust model as developed first by land trusts was extended to water rights through the Oregon Water Trust in 1993 and further extended in subsequent years to the Washington and Montana Water Trusts. These organizations have raised millions of dollars from members and interested foundations for the purposes of engaging in projects and transactions that return water rights to the public trust, that is, the water rights are dedicated instream either permanently or for a period as a lease. The Oregon Watershed Enhancement Board created by the state governor in 1996 as a means of heading off federal Endangered Species Act actions has funneled state funds (including revenues from vehicle license tags bearing a salmon insignia) to restoration efforts. Similar concern in the state of Washington over Endangered Species Act listings and other regulatory action prompted the state to begin providing its Department of Ecology with millions of dollars in public funds to undertake similar actions in 2000.

More recently, in 2003, the Bonneville Power Administration initiated a Columbia Basin Water Transactions Program to explore innovative strategies, including water rights transactions for environmental flows, as part of its obligations under the National Marine Fisheries Service Biological Opinion on the Columbia River System. In 2003 and 2004, the first two years of what was initially proscribed to be a five-year program, funds of up to \$2.2 million and \$4.2 million, respectively, were allocated. The Bio-Op calls for annual funding to reach at least \$5 million, which would be a significant portion of the larger BPA Fish and Wildlife Program responsible for expenditure of \$140 million annually. As administered by the National Fish and Wildlife Foundation, 11 local entities from Oregon, Washington, Montana, and Idaho have qualified to participate in the program. Although the funds are technically federal (BPA is effectively a federal parastatal agency), they are sourced from ratepayers, as BPA earns its revenues by producing and selling electricity in the Pacific Northwest states.

period) averaged 120–150 in each of the areas on annual traded volumes of 4,000 to 7,500 cubic meters. Subsequent increases in trading activity in the period 1997–98 in the GMID show 350 transfers and 25,000 cubic meters of permanent water trades, while there where 4,500 trades of 250,000 cubic meters of temporary water (Bjornlund and McKay 2002). Analysis confirms that the market for temporary water increases when allocations are tight (100%, for example)—up to 16% of the allocation was traded in 2001. In years with surplus water and large allocations (200%, for example), the percentage of water transferred temporarily is closer to 3–5% of allocations (Bjornlund 2002).

Price dispersion is noted in both markets, though it has decreased in the Murray over a ten-year period, suggesting a degree of market maturity (Bjornlund and McKay 2002). Prices for permanent transfers are consistently in the range of US\$ 0.70 per cubic meter (AUS\$ 0.99 per cubic meter) in that market. Temporary trades in 2000 and 2001 ranged from US\$ 0.03 per cubic meter (AUS\$ 0.04 per cubic meter) to US\$ 0.14 per cubic meter (AUS\$ 0.20 per cubic meter), depending on the balance of supply and demand (Bjornlund 2002).

Water exchanges now facilitate 10–40% of temporary trading levels in the three Murray-Darling Basin states (Bjornlund and McKay 2002). Further analysis of the GMID suggests that exchanges tend to be used for small and immediate transfers as well as transfers between distant parties, whereas private parties engaging in large, perhaps permanent and more local, transfers tend to prefer the business be done privately (Bjornlund and McKay 2002).

The temporary market can range from annual transfers to weekly transfers, allowing water users to access water when they need it and in response to timely information such as weather forecasts. In the temporary market, the market driver is water availability during the season, and the unit transferred is measured volumetrically; its delivery virtually assured (Bjornlund and McKay 2002). However, with permanent transfers, it is the water right, rather than the actual water, that is being purchased. Consequently, delivery is more uncertain and varies with the annual allocation determination. There are other limiting factors to the exchange as well, including alternatives and information requirements. The maximum price at which trades occur will be limited by commodity prices and alternative inputs; at some point, it is cheaper for farmers to buy grain than to buy water (Bjornlund 2002). Furthermore, the larger and more efficient an exchange is expected to be, the more that effective participation requires additional information inputs, which are not costless. (See Box 7.14 for a discussion of various pricing methods for water exchanges.)

There are specific conditions faced in the Australian case that may contribute to the inability of the exchanges to facilitate permanent transfers. An overall trend toward reducing industrial capitalization, particular tax benefits to leasing as opposed to purchasing or ''investing'' in water, the uncertainty over annual allocations and thus fluctuations in the value of water rights traded, and speculative interests create an environment in which the risk and volatility of permanent trades is too high to justify the investment (Bjornlund and McKay 2002).

7.3.2.2 Third-Party Impacts of Water Transfers

Third-party impacts include the full range of effects that a trade between a buyer and a seller have on other (''third'') parties and include impacts on other diverters of water on the canal or river, as well as environmental, social, and regional economic impacts (Gould 1988; Howe et al. 2003; Thompson 1997).

In the United States, impacts on other water users typically are accounted for through transfer regulations (Howe 1997). While the assessment of physical impacts is difficult, it is the degree to which more diffuse impacts are dealt with that can affect the workings of water markets (Gould 1988; Thompson 1997). Efforts to facilitate the regulatory process have led to only marginal increases in water transfers. More significant obstacles include inconsistent legal rulings, opposition by government entities that control water and conveyance systems, and concerns of communities that export water (Thompson 1997). The conventional view of water as strictly a public good that ought to be controlled only by the public (like the government) is at odds with a market approach and has hamstrung the development of permanent transfers (Thompson 1997).

Viewed from another perspective, the result of a study by the U.S. National Academy of Sciences concluded that third-party impacts were inadequately addressed by state rules (particularly for interest historically underrepresented in western water allocation) and remain the primary impediment to water transfers in the West (NRC 1992).

7.3.2.2.1 Environmental impacts

One perverse outcome of the implementation of water markets in Australia has been the activation of volumes of water that were

BOX 7.14

Pricing Methods for Water Exchanges and Banks

A key enabling factor in the success of water exchanges or banks is the design of the pricing mechanism. The pricing methods that have been used most predominantly are the sealed double bid and the bulletin board approaches (Bjornlund 2002). In the sealed double bid approach, buyers and sellers submit price per unit and quantity bids. The auction operator arrays buyers' bids in descending price order and sellers' bids in ascending price order. One clearing price is determined for the market, and this price is paid to all bidders such that no buyer pays more than she is willing and no seller accepts less than she is willing (Bjornlund 2002).

While the sealed double bid approach is expected to offer greater efficiency benefits than the bulletin board approach, in practice, it has not been as successful as expected. In its first year, the Southern Riverina (Australia) Irrigation District Council's Water Exchange tried the sealed double bid approach only to have participants request a bulletin board approach instead the next year. Since the exchanges have been preferred for temporary trades, the bulletin board approach offered more accurate market information and fluid, timely transactions (Bjornlund 2002). Since 1950, the Northern Colorado Water Conservancy District has maintained a bulletin board system in which short-term rentals are mostly traded (MacDonnell 1994). However, a downfall of the bulletin board approach is path-dependence—it matters who meets whom first, and therefore the market is not as efficient as it could be.

Other price discovery mechanisms that may be used in exchanges or banks include the following (Howe 1997):

- standing offers to buy and sell at fixed prices,
- sealed bid double auctions (and repeated sealed bid double auctions),
- live or sealed bid auctions for unique water rights,
- contingent water markets, including agricultural lease-outs. Participants enter contracts where the execution is contingent upon conditions at a certain time in the future. Two scenarios are negotiated and execution of the agreement always occurs because one of the scenarios (it rains, it does not rain) occurs. The bank would facilitate bringing together buyers and sellers, and the attraction to this would be that the transfers would not be permanent, and
- futures water markets.

previously unused (Isaac 2001). Seeking to dispel the ''markets are a panacea'' myth, Isaac points out that the doubling of water prices within four years denotes an improved efficiency of the market, but only in the purely economic sense that more water is now going to the highest and best use, including water that was not previously used (Isaac 2001). However, this would appear to be less a failure of the market than a failure of regulation. In setting the cap, regulators apparently did not apply a look-back period for beneficial use, as is done in the United States, to insure that the water being traded had actually been used for a beneficial purpose.

Implementation of an approach within a system where rights are privately held may facilitate the transfer of water to instream purposes according to the relative economic merits of water instream and out-of-stream. However, a ''free'' market in water is unlikely to be sufficient to reach environmental flow objectives, given the social and economic incentives that favor out-of-stream uses (Aylward 2003). In fact, the free-market Chilean experience bolsters this conclusion. In 1985, the Chilean government passed a separate law to subsidize the development of small- to medium-scale private irrigation projects, because the Water Code's market incentives had failed to stimulate private investment or water conservation initiatives (Bauer 2004). Instead, it is important to provide a regulatory framework that guides the reallocation of water between in- and out-of-stream uses in the direction desired by society (Bjornlund and McKay 2002).

The effectiveness of water markets in meeting water quality concerns is also tenuous. One study found that in the GMID, water was being sold from higher salinity soils to lower ones, though this did not hold true in South Australia along the River Murray. There, they found that due to changes in flow and dilution effects, salinity levels in the River Murray actually rose (Bjornlund and McKay 2002).

Much of the recent activity, at least in the United States, for acquisition of water rights instream has been for the protection of endangered fish species (Landry 1998, p. 6). Therefore, it is important to determine whether there is evidence that water quantity and water quality have improved as a result of market forces. In the 1991 California Water Bank experience, transfers were found to have both positive and negative environmental impacts, though they are not quantified in a cost-benefit analysis (MacDonnell 1994). Possible negative effects included damage to bird and wildlife forage and habitat as a result of removing grain crops, reductions in groundwater recharge from lack of seepage, reduction in groundwater quality, increased subsidence, and negative impacts on fishery conditions in the Delta caused by increased pumping. Potential positive impacts included improved surface water temperature, quantity, and quality, and reduced fish entrapment, though no conclusive evaluation was conducted on these effects (MacDonnell 1994). Even more surprising is the fact that measurement of the amount of water added to the system from market activities was imprecise.

7.3.2.2.2 Socioeconomic impacts

The underlying premise of using markets to allocate resources is that they result in efficient resource use. A study of the 1991 California Water Bank found that the Bank created net benefits of $91 million, including $32 million in net benefits to the agricultural sector (Howitt et al. 1992, in MacDonnell 1994). Studies in two watersheds in Chile also suggest significant net gains from trade as water moves from low to high value agricultural uses (Hearne and Easter 1995). The distribution of these economic impacts can be of concern, however, when they are concentrated in specific localities. Analysis of the California Water Bank suggests that total adverse impacts can be reduced by spreading acquisitions over a large geographic area (Howitt et al. 1992, in Howe 1997).

For permanent transfers, another concern is the extent to which a capital asset (water) is sold to finance recurring costs. In an Australian survey regarding the motivation of buyers and sellers to enter into transactions, it was found that in the Goulburn-Murray Irrigation District, only 26% of sellers used the revenue from sale of water to improve irrigation practices, while the rest used the funds for general revenue or for debt reduction (Bjornlund and McKay 2002). Similar results from the other districts echo the fact that the money was used mostly for income generation rather than reinvestment in the land.

In Chile's Limarí River Basin, the result of markets is that many small farmers have sold their water rights to larger farmers or to agribusiness corporations. However, little research has been carried out to assess whether this is an equitable result (Bauer 2004). In general, however, the Chilean experience suggests that the imposition of an unfettered free market in water into a developing context with significant existing socioeconomic inequality will lead to further inequities. (See Box 7.15.)

Thompson (1997) provides a response to these impacts by questioning why the transitional impacts illustrated above seem to be regarded as "special" in the water context. In general, in the U.S. context, resources are not constrained from moving to meet changing market demand. Federal or state assistance during such a transitional period is typically available on an economy-wide basis, and if such assistance programs are inadequate, it can be argued that they should be increased across the board, regardless of what causes the economic dislocation (Thompson 1997).

Other third-party impacts that have been considered with respect to water markets include the land use and associated impacts from transfers that lay land fallow. The California legislature has rejected legislation three times to require compensation for communities negatively affected by water transfers. Rather, the trend has been to voluntarily incorporate funds for local communities as part of transfer and land fallowing programs (Hanak 2003).

Effectiveness of market-based payment arrangements for delivery of water and watershed services will largely depend on stakeholder willingness to pay for them. This, in turn, depends on the level of confidence in the effectiveness of management actions to provide the ecosystem services and institutional arrangements needed to ensure access to benefits by those who pay the cost. These governance issues are covered in greater depth earlier in this chapter and important enabling conditions for markets of any kind.

7.3.2.3 Findings and Conclusions

A survey of global experiences demonstrates a range of water market tools used to reallocate water to new and higher value uses. Historically, the explicit purpose of water markets has been to improve resource efficiency, either within agriculture or among agricultural, municipal, and hydroelectric power needs. However, markets are increasingly coming to incorporate ecosystem needs as well. Since the mid-1990s, efforts have been made in the United States to explore the potential of water markets to meet ecosystem needs. However, markets have developed slowly and experienced limited activity, particularly of permanent transactions, due to public concern over the importance of local control of water resources (Thompson 1997).

The Australian experience shows the potential of markets to reallocate water to higher value uses, but cautions that instream needs are still mostly a public good, and therefore targets must be explicit and properly planned for in order to be achieved. Similarly, the Chilean experience demonstrates that a purely laissez-faire approach to creating markets for water may fail to protect the public good characteristics and have negative social and environmental consequences (Bauer 2004). Furthermore, the Australian experience demonstrates the need to prevent an increase in gross water use through the activation of previously unused water as markets develop.

In order for ecosystem needs to be achieved through markets, what is needed is either an explicit purchasing program for instream purposes or a system to reduce water allocations (a cap-and-trade mechanism whereby allocations can be ratcheted downward, for example). Experiences in the United States, including Oregon's Klamath Basin and Colorado's Arkansas River Valley, with allocation reductions imposed by regulatory action, demonstrate the conflict that can emerge when regulatory reductions are imposed, even if market mechanisms are used as a method for the redistribution of allocations (Howe et al. 1990). A more promising approach may be to provide direct governmental funding for buying back water rights to be retired instream, as implemented through the Bonneville Power Administration in the U.S. Pacific Northwest and considered in the Australian case.

Because the benefits to improved stream flow and freshwater ecosystems are still inherently public goods, the role of good governance and complete property rights for water remain fundamental enabling conditions for well-functioning markets. While there is a role for the use of markets to develop efficient water allocations, there is also a role for governments to regulate in providing stable and appropriate institutions for these markets to operate (Johansson et al. 2002).

BOX 7.15

Third-party Impacts of Water Markets on the Chilean Poor
(Bauer 2004)

There are a number of reasons why the Water Code and water markets in Chile have harmed poorer farmers. First, the military government did not provide the public with information, advice, or help in adjusting to the new law. Peasants and small farmers often learned about the new rules and procedures for acquiring or regularizing water rights too late to take advantage of them or to adequately protect themselves. Even in cases in which poor farmers got to know about the procedures, they were unable to use them without legal, financial, and organizational assistance.

Second, poor farmers are generally unable to participate in the water market except as sellers (if they are fortunate enough to have legal title to water rights, which is uncommon). They lack the money or credit needed to buy water rights. Their main hope for access to additional water is to benefit from the increased return flows that could result from improved irrigation efficiency on the part of more prosperous irrigators upstream. However, downstream users have no legal claim to such unused surplus flows, which are therefore an unreliable and insecure source of water.

Third, peasants and small farmers lack the economic resources and social and political influence needed to defend their interests effectively in the current laissez-faire regulatory context, in which private bargaining power is crucial. This is a disadvantage in two areas: conflicts over water use and conflicts over regularizing water rights titles.

7.3.3 Payments for Watershed Services

Economic incentives used to insure the delivery of watershed services essentially consist of payments to landowners to alter land management practices in the expectation of downstream benefits. A review completed in 2002 identified 63 examples from around the world of the application of market-based approaches to the provision of watershed services (Landell-Mills and Porras 2002). Key services paid for have included ensuring regular flows of water, protection of water quality, and control of sedimentation.

The types of arrangement vary depending on the characteristics of the service, the scale of relevant ecosystem processes, and the socioeconomic and institutional context. These range from informal, community based initiatives to more formal contracts between individual parties, and to complex arrangements among multiple parties facilitated by intermediary organizations. They may also include a mix of market instruments, and regulatory and policy incentives that are more likely to become necessary at larger scales, when threats are beyond the response capacity of individual communities, and a common set of rules may be required for purposes of trade among a larger and more diverse group of actors (Rose 2002). The instruments typically used to pay landowners for improving land management and protecting watershed services include transfer payments between governments and landowners, tradable development rights, voluntary contractual arrangements, and product certification and labeling. (See Box 7.16 for examples of each of these payment mechanism.)

The use of cap-and-trade mechanisms in managing upstream-downstream pollutant and water quality issues, as well as in wetland mitigation is also increasing. The assessment below focuses on transfer payment schemes. These schemes are perhaps the most straightforward approach for a government to take in providing economic incentives to landowners consistent with achieving watershed management objectives. They also offer the advantage of being susceptible to implementation in a comprehensive fashion covering large areas (or countrywide) and, therefore, have seen the widest application to the provision of watershed services and land management more generally.

7.3.3.1 Effectiveness

A key issue with respect to these upstream efforts at providing freshwater and other ecosystem services is whether the payment promotes an activity that actually produces the intended end result downstream. The ability to demonstrate this is key to building stakeholder confidence and willingness to pay for services. However, given the complexity of watershed processes, it is also a key challenge.

In the tropics, the general presumption that maintaining forests or reforestation is a superior land use, not just for watershed services, but for biodiversity, carbon sequestration, and other ecosystem services, often has been sufficient to launch payments for services programs. Thus a main concern has been to find buyers able and willing to pay, rather than what is needed, to insure they are provided, in a specific context (Pagiola 2002; Landell-Mills and Porras 2002). Unfortunately, when it comes to the tropics, the linkage between forest cover and downstream hydrological function is not a simple matter (Bruijzneel 2004, Bonell and Bruijnzeel 2004). Adding in the complexity of the linkage between hydrological function and the ecosystem services that contribute to the economic well-being of society makes it even more challenging to design effective interventions (Aylward 2004).

Recent reviews of the scientific knowledge base that has supported many watershed payment initiatives in Costa Rica suggests that management is limited by the lack of reliable and precise

BOX 7.16

Payment Mechanisms for Watershed Services

Transfer Payments: Transfer payments tend to be used at the national level, or over large areas and heterogeneous conditions. The best-known examples are the US Conservation Reserve Program and similar initiatives in some European Union countries. Under these programs, farmers are compensated for conservation measures based on a number of criteria, which include water quality and soil conservation. It is also the approach used in the Costa Rica FONAFIFO program, which provides payments to forest owners for multiple environmental services. Because of the broad social benefits associated with multiple services, these are usually supported by taxes rather than user fees, but may still pool funds from a variety of sources. In another example, in New South Wales, Australia, associations of farmers purchase salinity credits from the State Forests Agency, which in turn contracts with upstream landholders to plant trees, which reduce water tables and store carbon (Perrot-Maître and Davis 2001; State Forests of New South Wales 2001; Sundstrom 2001).

Tradable Development Rights: Tradable development rights require a strong planning and regulatory capacity to identify zones where development is to be restricted or permitted, and to enforce those restrictions for the period of the agreement. A well-known example in the provision of freshwater services in the United States is in the New Jersey Pinelands. The Pinelands Development Credit Program created credits for landowners whose land uses were restricted as a result of zoning. The credits may then be sold to developers in areas designated for development. The purpose of the program was to protect economic interests in the region as well as a very large drinking water aquifer (Collins and Russell 1988).

Tradable development rights are also used in wetland mitigation banks and conservation easements. Conservation easements involve the acquisition of water and land rights and can be implemented quickly and for any time period. It is one of many instruments used to implement the New York City Watershed Agreement, in which the city invests in upstream watershed protection measures to protect its water source, rather than build a new filtration plant (Perrot-Maître and Davis 2001).

Voluntary Contractual Arrangements: Voluntary contractual arrangements are often straightforward and may stand alone when negotiated among individual parties, such as the case of an agreement between the La Esperanza Hydropower Company and the Monteverde Conservation League in Costa Rica, which is the sole owner of the forested area upstream from the plant (Rojas and Aylward 2002). Agreements among numerous parties require that more consideration be given to the establishment of decision-making entities for purposes of allocating funds to priority conservation measures. An example is a trust fund, such as the Fondo del Agua (FONAG), established in Quito, Ecuador, to protect two upstream ecological reserves. This fund is overseen by a stakeholder board. It allocates pooled funds and "in kind" support received from municipal entities, NGOs, and private sources (Echavarria 2002).

Payments for watershed services are often referred to as market-based approaches, and are sometimes confused with privatization, but in fact usually consist of a hybrid package of instruments, both public and private, which accommodates the variability and uncertainty of services provisioning. For example, transfer payments may be made through voluntary contractual arrangements between government and landowners who provide services, using funds derived from user fees voluntarily paid by downstream water users. This set-up was used in Valle del Cauca in Colombia, where such an arrangement was necessary because the Colombian government had developed plans for watershed management, but did not have the ability to fund them. Funds were provided by associations of irrigation farmers who voluntarily paid additional fees to support the provision of a reliable water supply during the dry season (Echavarría 2002).

Product Certification and Labeling: Another mechanism is product certification and labeling. In this case, landowners are rewarded for specified management practices only indirectly—through the potential to use certification to increase their share of the market and possibly to obtain a price premium. Certification and labeling requires intermediary organizations to establish standards for labeling and to certify practices. An example is the Salmon Safe initiative, which certifies and promotes wines and other agricultural products from Oregon farms and vineyards that have adhered to management practices designed to protect water quality and salmon habitat. This tool has seen only limited application to specific watershed services, but has general applicability to improvements in land management, one typical output of which is improved watershed services.

information on forest water linkages (Pagiola 2002; Rojas and Aylward 2003). Instead, most are based on conventional wisdom, secondary sources of information, and selective references to literature reviews on forest hydrology. Regardless of the source material, they tend to invariably support statements that protection of forests will increase water yields (Rojas and Aylward 2003). In some cases, such as in the Arenal Basin (Castro and Barrantes 1998) and in Heredia (Castro and Salazar 2000), the values of watershed protection are calculated based on the opportunity cost of returning cleared land to forest cover, with no attempt to model and assess links between land use and hydrology, and to estimate the marginal values of water in specific consumption and production activities. Case studies from other regions also report the general absence of the scientific data needed to support valuation (Munawir et al. 2003; Geoghegan 2003; Rosales 2003; Echavarria 2004; The et al. 2004; Johnson and Baltodano 2004).

Given the diversity of the contexts in which payment arrangements are being developed and applied, and the impossibility of obtaining complete information that links causes with effects, and management actions with outcomes, a key consideration is whether a program is able to maintain the flexibility needed to make adjustments as barriers to implementation are encountered, and as lessons are learned. An example of an initiative that has been allowed to evolve over time, and continues to make adjustments, is the development of the Costa Rican FONAFIFO program, which is designed to support the multiple services provided by forests, of which fresh water is one. This program, which has been in place for five years, was built on an earlier ten-year program of payments for reforestation and the lessons already learned regarding barriers to implementation, as well as institutional arrangements already in place. Originally motivated by reduced timber supplies, the program had already made several adjustments over time, both to reflect the broader objectives of protecting natural forests and to allow greater farmer participation (Pagiola 2002). Additional adjustments were made in 2002 to include agroforestry and indigenous reserves in the program (Rosa et al. 2003).

A recent assessment of the social impacts of the program in the Virilla watershed found that this continuous institutional innovation has had significant benefits in terms of strengthened capacity for integrated management of farm and forest resources, and has contributed to the protection of 16,500 hectares of primary forest, sustainable management of 2,000 hectares, and reforestation of 1,300,000 hectares, which have had spin-off benefits for the protection of biodiversity and prevention of soil erosion. However, there are also high opportunity costs, particularly for smaller landowners, who tend to rely more heavily on small areas of cleared forest, and to combine forestry with other activities such as shelter for cattle and shade coffee. Those with larger tracts received a greater advantage from the program because they were able to maintain a higher proportion of the land in forest (Miranda et al. 2003).

Most initiatives have focused on links between upper watershed land use practices and downstream urban water supplies, sedimentation of hydropower dams, and irrigation canals. Although it has generally been found difficult to provide economic justification for interventions at this scale, little attention has been given to the more local level impacts within micro-watersheds, where land and water relationships can be better understood and stakeholders can be more directly engaged. Although the values placed on improvement of water quality are modest, it has been suggested that land use interventions for this purpose may be justifiable as part of an integrated community resource management strategy (Johnson and Baltodano 2004).

Because of the difficulties discussed, current trends are toward just such small-scale pilot initiatives that may have the potential to be scaled up to address problems at larger scales as capacity is developed. The PASOLAC program, which is engaged in 10 pilot initiatives, works to improve land and water management by small producers in the hillsides of Nicaragua, El Salvador, and Honduras, and is helping to develop markets for watershed services through the local municipalities. As of 2003, there were agreements in place between upland producers and downstream organizations in San Pedro del Norte in Nicaragua; Tacuba, El Salvador; and Campamento and Jesús de Otoro in Honduras (Pérez 2003).

This kind of bottom-up approach is generally expected to help insure that regional scale organizations are more representative of and accountable to local livelihood interests. Many current initiatives are also developing action research and learning approaches that can support capacity building as well as the exchange of knowledge (IIED 2004; RUPES 2004).

7.3.3.2 Findings and Conclusions

Given the heterogeneity and constant change in ecosystems and in human institutions, the site-specific nature of watershed processes, which are dominated by randomly timed and extreme events, and the difficulty of linking multiple causes and effects, or predicting outcomes, an adaptive approach to management is required, which implies the need for on-going assessment to support decision-making. However, preliminary assessments based on generalizations can provide some thumb rules and working hypotheses from which to begin. Perhaps the most significant challenge in such initiatives is to develop the capacity for a place-based approach to assessment, which is necessary to identify ecosystem functions that support provision of valued ecosystem services in their landscape context, and to select payment and institutional arrangements that are feasible and appropriate to that context. To be effective, market-based initiatives also need to be viewed as part of a long-term process of building appropriate institutions, and in the context of broader issues of structural reform and social change.

Effectiveness of market-based payment arrangements for delivery of watershed services will ultimately depend on stakeholder willingness to pay for them. This in turn depends on their confidence in the effectiveness of management actions for providing valued services, and that of institutional arrangements needed to insure access to benefits by those who pay the cost, both of which require greater attention.

The use of market arrangements does not automatically protect ecosystems, insure provision of their services, achieve an equitable distribution of costs and benefits, or reduce poverty. These objectives need to be made explicit and addressed in the design of economic interventions, which are as much an issue of governance as of economics. Intermediary organizations, governmental and otherwise, often play important roles in leveling the playing field and reducing barriers to market access by the poor. They can also help to reduce the transaction costs inherent in establishing arrangements among numerous stakeholders spread out over large and remote areas, and who may lack clear title to land, by facilitating agreements among them, negotiating on their behalf, and providing various kinds of legal and technical assistance, as well as through assessment and dissemination of appropriate information.

Given that land and water relationships are more detectable at the scale of micro-watersheds, it has been difficult to provide economic justification for interventions at the larger scales needed to insure the delivery of freshwater services to hydropower facilities and large municipalities. Emphasis is shifting toward much smaller-scale initiatives where causes and effects can be better understood and stakeholders may become more directly engaged. This provides a better point of departure for development of the capacity to respond to larger-scale problems in a way that is more representative of and accountable to livelihood interests.

7.3.4 Partnerships and Financing

In order to meet growing and as yet unmet demand for freshwater provisioning services in a sustainable fashion, a number of tasks related to physical infrastructure and the supplying ecosystems need to be performed. These tasks include building, operating, and maintaining infrastructure; information gathering; weather forecasting; managing ecosystems; controlling pollution; and preventing erosion. Unfortunately, most of these tasks, most of the time, are underfunded (especially in the developing world).

The use of economic incentives in managing terrestrial and aquatic ecosystems that provide fresh water are covered above. This section examines critical issues in the financing of water infrastructure, largely for domestic and industrial use. Investing in water resource development may produce enormous social returns (if it is well designed), but the financial returns are slow and low compared to the financial returns of investment in such other sectors as energy and telecommunications.

This section provides answers to the following questions: How has water infrastructure been financed? Why is water infrastructure financing prone to so many problems? What have been the trends in public-private partnerships? What are the binding constraints and enabling conditions that affect long-term financing of infrastructure needed to manage and use freshwater resources in a sustainable way?

7.3.4.1 Water Infrastructure Financing

Financing infrastructure means using funds to acquire long-term physical assets. The costs of developing and using water resources are eventually paid by either water users or taxpayers, or aid donors. However, financing can come from several sources. Table 7.5 shows the main sources for financing, the instruments or means they use, and their performance in the recent past. Summary information on financing sources for water infrastructure in 2002 suggests that 69% of financing is from the domestic public sector, 17% from external aid, and the remaining 14% from the private sector.

There is not much ground for optimism that these sources will increase their funding of water infrastructure in the future, considering that reliable long-term financing requires sources to be reimbursed. This reimbursement can only come from one of three groups: donors, taxpayers, or water users. The good news has come from the aid sector. Aid money is likely to increase in view of all the recent international agreements in development financing motivated by the setting of global social, economic, and environmental goals (the Millennium Development Goals, for example). At the 2002 United Nations Conference on Financing for Development in Monterrey, Mexico, donor governments and international agencies committed themselves to increasing their aid by 25%. However, the other two reimbursement sources (taxpayers and water users) will have more difficulties in rising to the challenge. Only a small proportion of local water systems recover their full operating and maintenance costs, let alone investment costs. Money from taxpayers and water users is not likely to increase unless the institutional problems that beset the fiscal and water sectors of most developing countries are solved.

Three features of the water sector that make cost recovery from water users very difficult. First, fixed costs constitute a very large proportion (around 90%) of total costs. This means that water agencies can operate (once the infrastructure is there) even with very low budgets. Second, water distribution is locally a natural monopoly (an industry in which technical factors—like the requirement of a network of pipes—preclude the efficient existence of more than one producer). Third, decisions on water tariffs are politically sensitive in most of the developing world. When taken together, these conditions imply that local water systems can operate (in the short run) with very low budgets, they do not face competition, and are politically constrained to implement changes that would allow them to recover costs.

The problem gets more complicated when funding is provided in foreign currency and the revenues are in local currency. Most of the loans that have to be repaid in foreign currency have suffered from this problem. Existing cost recovery contracts and financial instruments do not provide water systems with enough coverage to deal with this risk, especially in the case of massive devaluations, such as the peso devaluation in Argentina. These problems may beset water agencies whether they are operated by private companies or by public agencies, or by a mix of public and private organizations.

Furthermore, many national governments have devolved the responsibility for providing water services to regional or local agencies. However, these agencies usually have a very limited ability to raise finance on their own. Most of them need the support of the national government, as, for example, in guaranteeing loans. In some countries, sub-national levels of government are not allowed to raise money themselves. Even when they are allowed to do so, they tend to be short of expertise in financial management.

There are important constraints in fiscal resources, too. The fiscal weakness of many developing countries is the most obvious constraint, but not the only one. The fiscal relationship between the national and the sub-national levels of government is unclear and unpredictable in several countries. Under these conditions, long-term commitments are not likely to arise or be successful.

Problems have affected projects from the private and public sectors. Public scandals, accusations of corruption, tariff increases, allegations of failure to deliver the promised capital investments, and claims of failure to increase services to poor communities have plagued private sector participation in the water sector. However, the public water sector is not immune to these problems either. Allegations of failure to reform, improve efficiency and financial sustainability, limit political patronage, or expand access to as well as quality of services are common in many parts of the world. Unfortunately, the debate has been polarized between those who see private sector participation as a panacea and those who want to ban it completely because they think that the private sector cannot play a positive role in the water sector. These rapidly diverging views have increased the political risk and uncertainty associated with private sector participation and have led to an impasse among stakeholders on how to improve access to and the quality of water and sanitation services. The remainder of this section assesses the need, scope, constraints, and enabling conditions in order to better understand the potential role of private sector participation in the water sector, based on work undertaken by OECD (2003b).

7.3.4.2 The Need for Public-Private Partnerships

Governments around the world face difficult economic and political choices posed by the urban water sector. Securing safe, reli-

Table 7.5. Financing Sources and Means

Sources	Means	Observations
Water users	through tariffs or by developing their own infrastructure	lack of access to large volumes of finance constrain this source
Public water authorities and utilities	from user charges, loans, and subsidies	resistance to cost-recovering tariffs constrain this source
Private water companies	from user charges, loans, subsidies, and equity	resistance to cost-recovering tariffs constrain this source; the pool of companies has shrunk
National and sub-national governments	through subsidies, loan guarantees, and proceeds of bond issues	by far the largest source, but lack of coherent water strategies and weak fiscal conditions of national and sub-national governments in developing countries constrain it
Financial institutions (domestic, international, and multilateral)	offering loans	low private yields and long return periods discourage commercial lending; additionally, bank lending has declined because banks are more averse to lending to emerging markets; in some sectors (such as irrigation and hydropower), hostility to large storage projects has constrained this source
Bilateral and multinational aid agencies	through grants or soft loans	international aid for water and sanitation fell in the last few years (from $ 3.5 billion in 1996–98 to $ 3.1 billion in 1999–2001)
Local communities, self-help and nongovernmental organizations	through community participation, micro-credits, and application of low-cost technologies	knowledge gap (project design and financial aspects)

able, reasonably priced water and sanitation services for all is one of the leading challenges facing sustainable development. Many governments—local or national—have failed to recognize that, once it is piped, water for domestic and industrial use is a finite natural resource and an economic good. Instead, they have subsidized its use through a long history of underpricing and opposition to full cost recovery. Recently, the South African government has chosen to see domestic water as a basic right and to treat it as a public good, and is providing a base amount of domestic water free to the population at large.

Unfortunately, the long-term consequence of not setting a market price reflecting the cost of water provision and its true value to society has been a failure to recover costs, which, when combined with insufficient general funds, has led to water systems that are often operated inefficiently and, where services are unreliable, lack coverage, regular maintenance, and good design. With 88% of the 1.1 billion increment in global population through to 2015 likely to live in urban areas, there is a serious need to not only repair ailing systems, but build and operate new ones (WHO and UNICEF 2000).

Many towns and cities in developing countries have unreliable piped water systems and experience regular supply interruptions. Furthermore, the quality of services provided by existing systems is deteriorating, mainly because of the high capital costs of infrastructure, low user charges, and diminishing government resources for addressing urban water issues. The lack of investment in water supply and wastewater treatment threatens the quality of the services provided to citizens mostly in developing countries, provokes the decline in environmental and health standards, and contributes to poor demand management. Revenues and income for water companies are generally insufficient and unpredictable (OECD and World Bank 2003).

If the Millennium Development Goals on water and sanitation are to be met, current spending on water services of $75 billion a year needs to be increased to $180 billion (World Panel on Financing Water Infrastructure 2003). However, this target will be difficult to meet with public funds alone, as both government budgets and overseas development assistance have shown decreasing trends recently. Some governments are therefore increasingly looking to a range of private sector partners to provide access to two key resources: (1) improved management systems and technical options, and (2) private investment funds.

In OECD countries, investment needs also will increase substantially over the next few years, requiring greater efficiency through better management and the use of new sources for investments. For instance, in the European Union, about $5 billion per year are currently spent on water and wastewater services, and capital investment is predicted to increase by 7% a year for the foreseeable future (Owen 2002).

7.3.4.3 Public–Private Partnerships

Since the mid-1990s, an important approach that has been gradually introduced in the water sector is the notion of partnerships between public and private agents. Public-private partnerships correspond to any form of agreement (partnership) between public and private parties. These should not be confused with privatization, where the management and ownership of water infrastructure are transferred to the private sector. There is a wide range of approaches for involving the private sector as a partner in improving the performance of water and sanitation systems. (See Table 7.6.) Some options keep the operation (and ownership) in public hands, but involve the private sector actors in the management, operation, and/or financing of assets. A common point of all these options is that the government always retains responsibility for setting and enforcing performance standards—regardless of the form of private involvement chosen. The fact that the water sector is one of the natural local monopolies means that a strong regulatory role is required to insure that performance standards are met and the interests of consumers protected.

Table 7.6. Allocation of Public/Private Responsibilities across Different Forms of Private Involvement in Water Services. Cell shading in the table is to be interpreted as follows: dark grey = public responsibility; light grey = shared public/private responsibility; white = private responsibility. (Analysis of the authors based on OECD 2000)

Form of Involvement	Asset Ownership	Capital Investment	Design and Build	Operation	User Fee Collection	Oversight of Performance and Fees	Typical Duration
Design and construct contracts							—
Service contracts							1 to 3 years
Management contracts							3 to 5 years
Lease contracts							8 to 15 years
Joint ventures							
Build, operate, transfer							15 to 25 years
Concession Contracts							20 to 30 years
Passive public investment							—
Fully private provision							—

There is no universal "right answer" on how to use private investment to help improve water services. Ultimately, governments need to devise arrangements that fit the local context, and some may decide that public-only is best. Where the private sector is hesitant to engage itself, it might be suitable to start with methods that involve low risk for the private operator (as in service contracts), moving only later toward more ambitious forms of involvement, if considered appropriate (OECD 2003a). The degree of risk-sharing will be an important determinant of private sector participation.

The most commonly cited advantages of private sector participation are that it brings technical and managerial expertise to the sector, improves operating efficiency, entails injections of capital and greater efficiency in its use, and increases responsiveness to consumer needs and preferences. It is often assumed that the private sector has significantly better access than governments to capital flows and to the technical know-how that is essential in the provision of critical water services.

In some OECD countries, public-private partnerships have existed in the water sector for more than a century, as in France; and in most other OECD countries, they have existed for more than a decade. The options range from limited private investment to full divestiture, predominantly in England and Wales. (See Table 7.7.) Water supply in France is in public ownership, but management is a mix of public and private systems. The French municipal authorities act as an economic regulator. In the United Kingdom, the ownership and management are private, but the economic regulator (Office of Water Services) is an independent body. The United States has a part public and part private ownership structure, but is dominated nevertheless by the public sector. There is a growing tendency in OECD countries for water systems to be managed by groupings of municipalities so as to organize supply at a larger scale. Other forms of consolidation have also been occurring. An example of this is the case of the Netherlands, which reduced the number of water boards from 210 in 1950 to 15 in 2002 (van Dijk and Schwartz 2002). These developments are likely to attract additional private sector interest for water due to the increased project size and potentially associated economies of scale.

Table 7.7. Share of Public–Private Partnerships in Key OECD Urban Markets (OECD 2003b, based on BIPE 2001)

Country	Public Sector Management *(percent of population served)*	Private Sector Management *(percent of population served)*
Germany	96	4
France	20	80
United Kingdom	12	88 (100 in England)
The Netherlands	100	—
United States	85	15

In the developing world, while the 1990s saw a significant increase in private sector participation in the water sector it is estimated that only 3% of the population in poor or emerging countries is provided with drinking water through private operators (Owen 2002). As shown in Figure 7.2, since 2000, the number of projects has decreased substantially. In effect, the important peaks in certain years correspond to the large concessions that took place in Buenos Aires in 1993, in Manila in 1997, and the

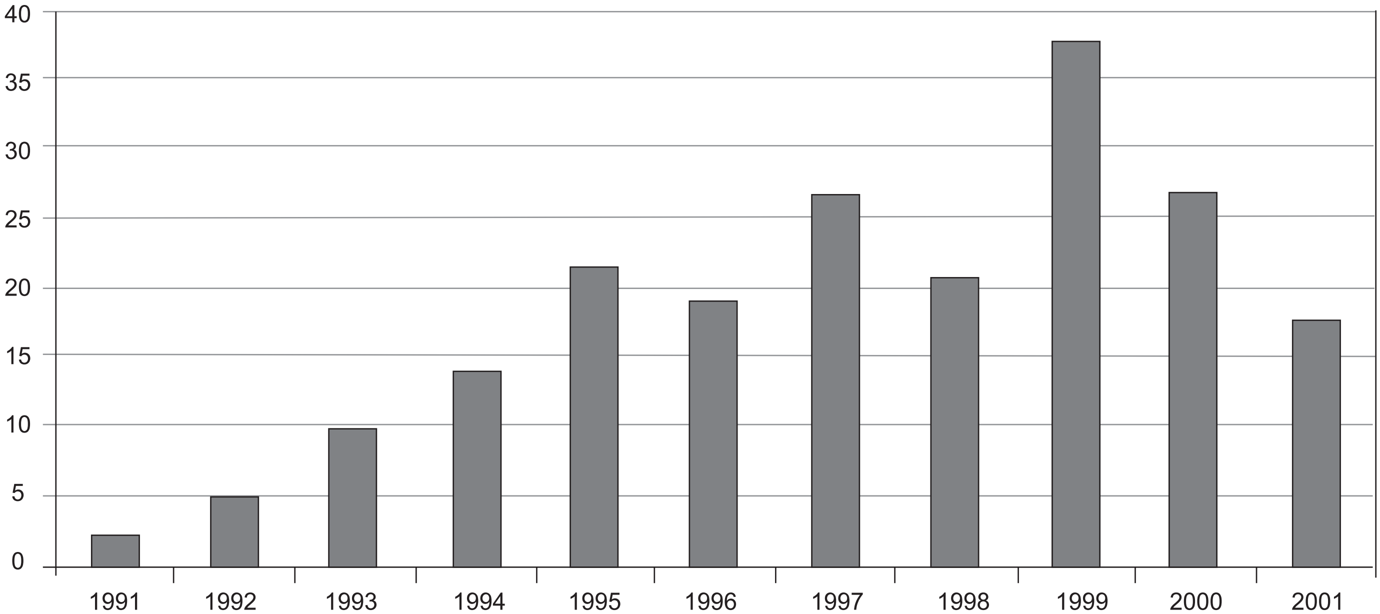

Figure 7.2. Number of Projects with Private Participation in Developing Countries, 1990–2001 (World Bank 2003)

privatization of the Chilean water utilities in 1999 (World Bank 2003b).

According to the World Bank, public-private partnerships are most common in Latin America, followed by East Asia and the Pacific, and Europe and Central Asia. While the nature of private sector participation may range from partial financing of investments to an increasing role in the operation of services, most countries have opted for the concession approach, in which the private sector participates in managing some services, but the public sector retains ownership of the system.

It is important to bear in mind that many examples of efficiently managed public water and sanitation utilities exist, and that the characteristics of the public sector differ among countries. Thus in many countries, it is not necessarily the public sector per se, but factors such as faulty incentive structures, politicization of appointments and management, and other bureaucratic weaknesses that contribute to poor performance. Despite widespread belief in the potential for efficient use of the private sector in some areas of service provision, empirical evidence of the relative merits of private and public management in the water sector is relatively limited (OECD 2003a).

7.3.4.4 Effectiveness

Recent experience with the involvement of the private sector mainly in non-OECD countries suggests that there are major obstacles that significantly hinder greater private sector participation in urban water services. Despite high hopes that private sector participation might help overcome the financing gap for achieving international goals for access to water and sanitation, an increasing number of water sector projects with private sector participation appear to be in crisis, often due to the difficult economic situation in the host country. The number of such projects has been decreasing and investment flows have been slowing over the last four years. This has triggered recognition by both public and private actors of a number of systemic problems in the design of projects, for which solutions need to be found. These include weak regulatory set-ups, lack of political support for private sector participation, need for long-term debt finance, low returns on investment, fragmented deal size, poor creditworthiness of local governments, poor contract and project design, and a frequently inappropriate allocation of risks between involved parties. Some of these key issues are highlighted below.

Regulatory frameworks in host countries are often insufficient and unstable. This generates significant uncertainty about future cash flows for the private operator, since essential cost elements (such as waste water treatment requirements) as well as revenues (such as tariffs) cannot be anticipated. This situation, together with the often weak levels of contract enforcement, is among the key reasons for the low use of public-private partnerships in many emerging market economies and developing countries. Technical assistance from donors can help to remove many of these obstacles by providing support for capacity building and institutional reform, but ultimately, political commitment is also needed (OECD and World Bank 2002).

Political commitment to public-private partnerships at all relevant government levels is essential, since water is perceived to be more than a simple good by both consumers and many politicians. This has sometimes been overlooked, leading to the rapid loss of political backing as soon as the projects encountered initial difficulties (OECD and World Bank 2002).

Networked water systems have extremely high capital costs, well in excess of other infrastructure services. They are mostly financed with debt, for as long a term as is commercially available. Given the high initial costs, extremely long pay-back periods are necessary, and it is essential that revenue streams be as secure as possible. Urban water services are also a business with relatively low rates of return on investment. Due to these sectoral specificities, private operators are particularly sensitive to the quality of the investment climate and the level of risk, which is an important obstacle to public-private partnerships in many regions of the world. Furthermore, in the last couple of years, the risk aversion in the infrastructure market in general has increased because of several events, including the September 11, 2001, attacks, the recent corporate bankruptcies, the reduced number of strategic investors, and the rating downgrades (OECD and World Bank 2002).

Finally, many public-private partnerships have encountered difficulties due to insufficient attention being paid to the social consequences of involving the private sector as they often implied tariff increases due to a move towards the full recovery of operation and maintenance costs through tariffs. Another reason is the popular mistrust of institutions involved in such projects. Unless continued access to water services of the poorest sections of the population is insured at a reasonable cost, and sufficient levels of transparency in decision-making insured, major social resistance must be expected to public-private partnerships. Making sure that social protection schemes are being developed prior to or in parallel with public-private partnerships is therefore a crucial success factor (OECD and World Bank 2002).

Even if these obstacles are overcome, it must be recognized that such partnerships are not a panacea. Public-private partnerships involving international private sector operators cannot solve all the problems in the water sector, nor can they be applied everywhere. Clearly, the private sector will only operate where certain profitability requirements can be met, which considerably limits the scope for public-private partnerships.

First, for some of the reasons mentioned earlier, major investment in such partnerships in the water sector is likely to focus on OECD and emerging market economies, where the environment for foreign investors is most favorable. This has been so in the past and is unlikely to change significantly in the future. Most of the applications have been in high- and middle-income countries, leaving least developed countries uncovered. For instance, less than 0.2% of all private sector investments in the water and sanitation sector of developing countries went to sub-Saharan Africa (United Nations Millennium Project 2003).

Second, there are only a limited number of international water operators, and their human and financial capacities allow for the management of only a limited number of projects. The three largest private operators account for more than 50% of the global market. Public-private partnerships in non-OECD countries, therefore, focus on urban areas that are likely to yield the most substantial revenue flows and offer the best opportunities to achieve significant economies of scale—typically large cities with populations of 500,000 or more.

While the potential arena of operation of international private operators is limited, opportunities for the involvement of new entrants may exist. This is particularly the case of domestic private sector companies in developing countries. The mobilization of these actors may help enlarge the scope of public-private partnerships.

7.3.4.5 Findings and Conclusions

There is a clear mismatch between the high social value of freshwater services for domestic use and the resources that are being allocated to manage water. Insufficient funding to expand water infrastructure is one manifestation of this mismatch. Both inherent characteristics of the water sector (high fixed cost, low returns, long pay-back periods) as well as institutional problems (political interference, inadequate legal frameworks, poor management structures) explain the gap in funding infrastructure. No single source will be able to bridge this gap on its own. There are several sources of funding water infrastructure and all have a role to play. In addition to a more creative use of existing financial instruments and the development of new ones, changes at different levels are needed in order to unleash financing sources. At a national level, legal frameworks have to provide more certainty to the parties of long-term commitments. The water sector has to establish its priorities in a clear way and produce programs that include the definition of financing needs and sources. Finally, at the agency level, cost recovery must be improved and managerial and technical capacities, enhanced.

In the future, public-private partnerships should take into account the following priorities. Governments should be clear on their strategies and priorities for the water sector, and plan accordingly. There must be an effort to optimize the use of existing financial vehicles and introduce new ones. Finally, a long-term sector strategy should be adopted in order to achieve more efficient urban water management (OECD and World Bank 2003).

The enabling conditions for adequate long-term financing of the infrastructure needed to manage and use freshwater resources in a sustainable way go well beyond the conditions of financial markets. Apart from having access to resources, the financial strength of the water sector requires clear and transparent priority setting as well as developing programs to meet these priorities. Some actions that would facilitate these tasks include establishing priorities as well as service standards in a transparent and clear way, developing and implementing programs and actions to meet those goals, and obtaining access to the resources that will allow the implementation of programs and actions.

7.4 Supply Infrastructure and Technologies

As discussed earlier, the demands for fresh water have grown drastically over the last few centuries and provided the stimulus for the emergence of physical infrastructure that regulates the natural flow characteristics of free-flowing rivers. In terms of infrastructure for the enhancement of storage, a large number of dams have been constructed all over the world. ''In North America, Europe, and the former Soviet Union, for example, three-quarters of the 139 largest river systems are strongly or moderately affected by water regulation resulting from dams, inter-basin transfers, or irrigation withdrawals'' (Gleick et al. 2001, p. 22). In addition, hundreds of thousands of kilometers of dikes and levees have been constructed with the purpose of river training and flood protection. While these structures have clearly provided increased supply of fresh water for many uses, as well as flood control, all too often, they have had debilitating effects on the surrounding ecosystems, their naturally occurring services, and their biodiversity.

A number of other well-developed and documented technologies are available for improving efficiency and water resource management (World Commission on Dams 2000):

- micro-watershed level conservation of rainwater through physical or vegetative land management;
- rooftop rainwater harvesting;
- water recycling and reuse;
- desalinization for domestic water supplies (in coastal areas);
- on-farm agricultural water conservation, such as sprinklers and drip irrigation;
- crop selection and irrigation management based on meteorological conditions;
- improving crop productivity (with same or lower water use) through technological inputs;
- improvements to irrigation and municipal system management and conveyance;
- household water-saving devices, such as low-flow showerheads and water efficient toilets;
- managed flood releases from reservoirs to simulate historic flooding and impacts on downstream landscapes and ecosystems; and
- improved reservoir management and technologies for reducing evaporation loss.

Another technology that may come into play in coming decades is that of inter-basin transfers, particularly large, mega transfers between major river systems. For example, in India and

China, transfer projects costing hundreds of billions of dollars are proposed. These projects do not recognize ecosystem water needs and downstream consequences (Bandyopadhyay and Perveen 2004).

In this section, brief assessments are presented of three very different response options from the infrastructure and technology field: large dams, wetland restoration and mitigation projects, and desalination. To some degree, these represent the predominant past approach to water development, a current ecosystem approach to ecosystem restoration (or at least maintenance), and a promising new supply technology

7.4.1 Large Dams

A common response to water supply augmentation is the construction of large dams, which are defined by the International Commission on Large Dams as those with a height greater than 15 meters from the foundation or those that are 5–15 meters high with a volume of more than 3 million cubic meters. More than 45,000 large dams exist worldwide (WCD 2000).

Large dams can be used to regulate, store, and divert water for agricultural production and consumptive use in urban and rural areas. They were seen as integral components of the Green Revolution and were promoted widely in this period. Over half of the world's large dams have been built for irrigation and water supply purposes. Beyond water supply augmentation, large dam use also includes hydroelectric power generation and flood control (WCD 2000).

Large dams have proved especially useful for providing greater security in the face of water scarcity and variable supplies of water, a feature of countries with semi-arid catchments, where flow is highly variable and characterized by periodic drought such as South Africa, Spain, and Australia (WCD 2000).

7.4.1.1 Effectiveness

A full assessment of a few trillion dollars worth of infrastructure is beyond the scope of the current assessment. Instead, the experience with large dams is briefly summarized, based on the results of the World Commission on Dams, a multistakeholder international assessment, which recently spent three years and over $10 million dollars to carry out just such a task.

The benefits attributed to large dams include water supply to growing populations; increased food production; electric power for domestic, industrial, and other uses, as well as navigation and flood control. However, the environmental and social impacts of large dams are also well-known and have led to the very controversy and stalemate that resulted in the call by different parties to the debate over dams for an independent commission.

The WCD report identified a number of central issues in the dam debate, including: performance (costs and benefits), environmental impacts and sustainability, social impacts and equity, economics and finance, and governance and participation.

Of particular relevance to ecosystem health and human well-being are the environmental, social, and economic issues raised in the report. Environmentally, the impact of large dams on freshwater ecosystems is widely recognized as being more negative than positive. The impacts include, amongst others: changes in flow and sedimentation patterns; irreversible loss of species and populations, such as upstream and downstream fisheries; loss of habitat and associated biodiversity and ecosystem services from floodplains, wetlands, and estuarine and marine ecosystems; and greenhouse gas emissions from decaying organic material in the flooded basin.

The direct social impact of large dams is striking—they have led to the displacement of 40–80 million people worldwide and terminated access by local people to the natural resources and cultural heritage in the valley submerged by the dam (WCD 2000). The perennial freshwater systems established by large dams also contribute to health problems. For example, epidemics of Rift Valley fever and bilharzias coincided with the construction of the Diama and Manantali dams on the Senegal River (World Bank 2003a). Aside from the direct impacts of large dams, the benefits of their construction have rarely been shared equitably—the poor, vulnerable, and future generations are often not the same groups that receive the water and electricity services and the social and economic benefits from dams (WCD 2000).

Large dams were also found wanting from an economic and financial perspective. Pre-construction studies are typically overly optimistic about the benefits of projects and underestimate the costs. In a sample of 248 dams compiled by the WCD, the average cost overrun was a full 50% of the originally estimated costs. Further, the simplistic economic cost-benefit analyses applied often fail to adequately integrate the social and environmental impacts into the planning cycle.

As a result of these findings, the WCD concluded that, "The positive contribution of large dams to development has, in many cases, been marred by significant environmental and social impacts which are unacceptable when viewed from today's values" (WCD 2000, p. 198). The construction of large dams remains a viable option for augmenting water supply, but the conclusions of the WCD report suggest that the large dam option is one that needs to be carefully examined given past experience.

An underlying principle in the WCD approach is the recognition of stakeholder rights and a negotiated decision-making process. The publication of the *Report of the World Commission on Dams* served to bring to light many of the social and ecological costs of major water infrastructure. The report implicitly suggests the need to replace the traditional engineering view of dams and development with a new and more widely acceptable approach (Bandyopadhyay et al. 2002). The core of this approach is based on the newfound strategic importance and economic significance of the ecosystem services provided by rivers and watersheds.

7.4.1.2 Findings and Conclusions

The construction of large dams started in the early twentieth century and peaked during 1960–70. For various reasons, including resistance by people's movements, changing trends in project finance, and growing concern over the environmental and social impacts, the level of dam construction during 1990–2000 fell to almost the level of all the dams constructed during 1901–50. Many proposed dams have been postponed or canceled in the last two decades (Gleick et al. 2001, p. 22). Since the WCD process concluded in 2000, a number of new large dams have moved forward or been proposed (Three Gorges in China, and Kárahnjúkar in Iceland, to name two), and a number of large projects have run into difficulties (such as Bujugali in Uganda and Nam Theun II in Laos). Given the long lead time necessary to plan, finance, and build large dams and the continued controversy over the WCD report, it is too early to say what impact the report will ultimately have on the future of large dams. All that can be said at this point is that the report has been widely circulated and discussed, and that it has enhanced the legitimacy of the position that continued reliance on a supply-driven engineering approach is not sufficient to overcome the challenges ahead (Bandyopadhyay et al. 2002).

7.4.2 Wetland Restoration and Mitigation

Wetland restoration is a broad response category. In a strict sense, wetland restoration refers to the process of inducing and assisting the abiotic and biotic components of an ecosystem to return to their original state (Bradshaw 1997). However, definitions of restoration also encompass (1) actions to improve the condition of a site that may not necessarily be in the direction of pre-existing conditions (Bradshaw 1997), and (2) mitigation projects that seek to create wetlands to replace those that may have been lost from human interventions (Zedler 2000). This assessment considers wetland restoration in its broadest sense and is thus consistent with the Ramsar Convention definitions of this term.

Wetland restoration approaches are numerous and include engineering solutions such as backfilling canals and the removal of contaminated groundwater, biological interventions including controlling the impact of feral fish and reestablishing wetland plants, through to hydrological management to increase the effective inundation across floodplains and reintroduction of drying cycles.

7.4.2.1 Effectiveness

Mitsch et al. (1998) suggest that wetland restoration has become controversial in part because of the uncertainty about what is necessary to create and restore wetlands, that is, what combination of processes leads to the establishment of a desired combination of wetland structure and function. Such understanding would include the germination requirements, seed viability, and seedling growth characteristics of target wetland plants (van der Valk et al. 1999). However, imperfect knowledge regarding ecosystems can produce unexpected outcomes, as demonstrated in the uncertainty shown in models used to predict changes in wetland structure and function (Klotzli and Grootjans 2001).

With respect to the restoration community's understanding of the influential factors, it can be confidently said that the outcome of restoration projects is influenced by variables such as landscape context and site selection, hydrological regime, the rate of development of ecosystem attributes, nutrient supply rates, disturbance regimes, seed bank condition, invasive species, and life-history traits (Zedler 2000).

One of the major disagreements among wetland scientists is in relation to the role of abiotic conditions, especially hydrology, versus life-history traits in determining wetland structure and function. Mitsch et al. (1998) suggest that restoring the hydrological regime, or more generally abiotic conditions, is sufficient to reestablish structural features, particularly vegetation (the self-design approach). This is in contrast to Galatowitsch and van der Valk (1996), who suggest that at least in prairie potholes, dispersal is likely to be more limiting.

These approaches have implications for the restoration techniques applied—self-design is likely to focus on recreating hydrological features where the design approach will see engineering and replanting strategies as important. While this represents a major area of debate, it does not undermine wetland restoration as a response to wetland degradation, rather it places greater emphasis on understanding the factors that limit the rehabilitation of a site.

Perhaps of greater importance than the process of restoration is the actual restorability of a site. This has been highlighted in recent studies with the finding that it simply may not be practical to restore some wetlands because of the extent of degradation. This is an issue predominantly relating to abiotic factors such as wetland soils, the composition of which may have been irreversibly altered through changes in pH and nutrient status. Even if the changes are reversible, the time taken might be decades to centuries (Zedler 2000).

Cost is also an issue for wetland restoration and will reflect the extent of degradation and the objectives for restoration. Where the drivers of degradation operate at a local scale and are easy to identify and rectify, the cost may only be in terms of the voluntary time and effort provided by community groups. In contrast, where degradation is due to a multitude of factors operating at a regional or catchment scale, the cost may be in hundreds of millions of dollars (as in the $685 million Florida Everglades restoration project; Young 1996). This is certainly the case for large-scale environmental flow projects that aim to restore large wetland areas.

The success of wetland restoration is ultimately determined by the ability of a project to meet its original goals. In this regard, it has been noted on a number of occasions that wetland restoration projects suffer from poorly stated or unstated goals and objectives (Zedler 2000). An example of a poorly stated goal is one that is too generic to meet the original intent of the project, for example, the increased diversity of wetland plants within five years. This may be achieved with little movement toward reference conditions or more detailed aspects of structure and function.

The setting of well-stated goals must form part of a broader, comprehensive, and rigorous process for planning, developing, implementing, and evaluating restoration projects. This is widely accepted among wetland management practitioners as a key to success. In this regard, numerous frameworks for designing wetland restoration projects have been articulated, much of which has been synthesized in Ramsar's *Principles and guidelines for wetland restoration* (www.ramsar.org). Consistent with these frameworks is an adaptive management approach discussed in more detail in Chapter 15 of this volume, which allows for iterative learning and a chance to build on imperfect knowledge.

Although there is no definitive answer as to the ingredients of a successful wetland restoration project, it can be said with high confidence that there is a positive correlation between successful restoration and (1) a clear identification of the drivers of degradation, (2) where a small number of drivers are active, (3) when drivers operate at a local scale, (4) where drivers are inexpensive and easy to mitigate, (5) where the trade-offs required to mitigate the drivers are minimal, and (6) where the degradation of the wetland is reversible.

Despite meeting some of these criteria, where functional equivalence to an original state is the goal of restoration, success may be difficult to measure. For example, Mitsch and Wilson (1996) reported that where goals are specific and relate to aspects of wetland functioning, the time required to measure success could be 15–20 years. This is consistent with van der Valk's (1981) work on prairie pothole wetlands indicating that succession in wetlands may occur in 25-year cycles. Mitsch and Wilson (1996) suggest that in the case of restoring or creating forested wetlands, coastal wetlands, or peat-lands, it may require even more time.

7.4.2.2 Findings and Conclusions

Achieving functional equivalence is important when creating new wetlands to replace those that are destroyed. The question asked often is: "Does the structure and function of the new wetlands replace that of the old?" As for restoring degraded natural wetlands, this is hampered by uncertainty in the role of different abiotic and biotic factors and the observation that each wetland is a product of the unique contributions of these factors. The conclusion of numerous studies is that created wetlands rarely perform the same functions or house the same biodiversity as the original site.

For this reason, it is unlikely that created wetlands are going to structurally and functionally completely replace destroyed wetlands. This may be equally the case for degraded natural wetlands and is reflected by the Ramsar Convention, which notes that "the maintenance and conservation of existing wetlands is always preferable and more economical than their subsequent restoration" and that "restoration schemes must not weaken efforts to conserve existing natural systems."

7.4.3 Desalination

Desalination is the production of fresh, low-salinity potable water from saline water source (seawater or brackish water) via membrane separation or evaporation. The mineral/salt content of the water is usually measured by the water quality parameter total dissolved solids in milligrams per liter or parts per thousand. The World Health Organization and the U.S. Environmental Protection Agency, under the Safe Drinking Water Act, have established a maximum TDS concentration of 500 mg/L as a potable water standard. This TDS level can be used as a classification limit to define potable (fresh) water. Typically, water of TDS concentration higher than 500 mg/L and lower or equal to 15,000 mg/L is classified as brackish. Natural water sources such as sea, bay, and ocean waters that usually have TDS concentration higher than 15,000 mg/L are generally classified as seawater. For example, Pacific Ocean seawater along the U.S. west coast has a TDS concentration of 33,500 mg/L, of which approximately 75 % is sodium chloride.

Approximately 97.5% of the water on our planet is located in the oceans and, therefore, is classified as seawater. Of the 2.5% of the planet's fresh water, approximately 70% is in the form of polar ice and snow, and 30 % is groundwater, river and lake water, and air moisture. Even though the volume of Earth's water is vast, less than 10 million of the 1,400 million cubic meters of water on the planet are of low salinity and are suitable for use after applying conventional water treatment only. Desalination provides a means for tapping the world's main water resource—the ocean.

7.4.3.1 Effectiveness

By 2004 over 17,000 desalting units with a total production capacity of 37.75 million cubic meters per day (10 billion gallons per day) have been installed in approximately 120 countries (Wangnick 2004). Desalination techniques predominantly use either thermal or membrane processes (Buros 2000). Thermal desalination technologies use a variety of forms of distillation, including multiple effect distillation, multistage flash distillation, or vapour compression distillation. Membrane separation is typically accomplished by reverse osmosis or electrodialysis technologies. (See Box 7.17.)

Most of the large seawater desalination facilities built in the past 10 years or currently undergoing construction are delivered under public-private partnership arrangement using build-own-operate-transfer method of project implementation. The BOOT project delivery method is preferred by municipalities and public utilities worldwide because it allows cost-effective transfer to the private sector of the risks associated with the number of variables affecting the cost of desalinated water, such as: intake water quality and the often difficult to predict effects on plant performance; permitting challenges; start-up and commissioning; fast-changing membrane technology and equipment market; and limited public sector experience with the operation of large seawater desalination facilities (Voutchkov 2004a).

BOX 7.17

Reverse Osmosis: Removing Salt using Semi-permeable Membranes

Reverse osmosis separates solutes from saline water by forcing it through a semi-permeable membrane under pressure. Unlike distillation no heating is required with most of the energy used to pressurize the saline feed water. The basic system components are pre-treatment, high-pressure pumps, membrane assembly, and post-treatment. Beyond its use for desalting, reverse osmosis can also be used for removal of other impurities and contaminants such as iron, lead, nitrate, endocrine disruptors, arsenic, disinfection by-products, bacteria, viruses, and other pathogens and emerging contaminants. The first RO membrane was first developed at the University of California at Los Angeles in the early 1960s by Loeb and was used to produce drinking water from seawater. This relatively thick membrane was made from cellulose acetate and required feed pressures in excess of 2,000 pounds per square inch (psi). Today RO membranes used to desalinate seawater are made of thin-film composite plastic materials and require about 800 to 1,200 psi, while brackish water applications may necessitate feed pressure ranging from 100 to 600 psi. The feed pressure required depends primarily on the total dissolved solids (TDS) concentration and the temperature of the water—lower TDS levels and warmer waters requiring lower feed pressures.

The largest RO membrane desalination plant worldwide in continuous operation today is located in the United Arab Emirates (Taweelah) and has capacity of 227,000 cubic meters per day. The largest plant in construction is the Ashkelon seawater desalination facility in Israel; this plant will be operational in the spring of 2005 and will have capacity of 395,000 cubic meters per day. The Yuma desalting plant in Arizona is the largest U.S. reverse osmosis brackish water desalination plant and can produce about 275,000 cubic meters per day. The largest seawater desalination plant in the United States is the Tampa Bay, Florida, facility, which has potable water production capacity of 95,000 cubic meters per day.

Up until the 1970s, desalination was predominately performed using distillation techniques, with some commercial units capable of producing up to 8,000 cubic meters per day. Subsequent technological improvements have seen an expansion in the use of membrane processes, especially reverses osmosis. Currently, multistage flash distillation only accounts for 36.5% of total installed desalination capacity worldwide, down from 51.3% a decade ago. By contrast, plants using RO have risen from 32.7% in 1993 to 47.2% today (Wangnick 2004). The increasing popularity of RO membrane desalination is driven by remarkable advances in the membrane separation and energy recovery technologies, and associated reduction of the overall water production costs.

Other less commonly used techniques for desalting water include ion-exchange methods, freezing, membrane distillation, and solar and wind driven systems (Buros 2000). Solar systems include solar stills, which heat and vaporize water from a ground level basin and then collect vapor from a sloping glass roof. This technique faces drawbacks such as high capital costs, vulnerability of glass to weather damage, and the large collection areas required (Gleick 2000). An alternative to these systems is to use wind and solar generated electricity to drive more traditional desalting processes.

The developments in seawater desalination technology during the past two decades, combined with transition to construction of large capacity plants, co-location with power plant generation facilities and enhanced competition by using the BOOT method of project delivery have resulted in a dramatic decrease of the cost of desalinated water. Recent large reverse osmosis desalination projects in the United States, Israel, Cyprus, Singapore, and the Middle East have installed costs of approximately $0.50 per cubic meter, down from $1.50 per cubic meter in the early 1990s (Voutchkov 2004b).

7.4.3.2 Findings and Conclusions

There is no single best technique for desalination. The selection of a desalination process depends on site-specific conditions, economics, the quality of water to be desalinated, the purpose of use, and local engineering experience and skill (Gleick 2000). Operating and capital costs in particular are influenced by the capacity and type of desalination plant, the quality of feed water and the energy required to drive the process (Buros 2000). Water from desalination is generally expensive, although costs have decreased in recent years to such an extent that in some areas of the United States desalting brackish water is cheaper than alternative measures such as piping conventionally treated water (Buros 2000).

Although, no major technology breakthroughs are expected to bring the cost of seawater desalination further down dramatically in the next several years, the steady reduction of desalinated water production costs coupled with increasing costs of water treatment driven by more stringent regulatory requirements, are expected to accelerate the current trend of increased reliance on the ocean as an environmentally friendly and competitive water source. This trend is forecasted to continue in the future and to further establish ocean water desalination as a reliable drought-proof alternative for many communities in the United States and worldwide.

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