2 Water Pricing in Irrigation: Mapping the Debate in the Light of Experience

F. Molle and J. Berkoff

Introduction

This chapter provides a broad discussion of water pricing in agriculture, scrutinizes arguments sequentially, gives examples from the literature and indicates links to other chapters. It suggests the conditions under which water pricing is likely (or not) to bear fruit, and assesses its potential for alleviating the global and local water crises. The focus is on public large-scale gravity schemes although groundwater and communal systems are also referred to, albeit in less detail.

Charging for water use or disposal is not an end in itself, but an instrument for achieving one or more policy objectives (Fig. 2.1). A water charge may be *a financial tool* aiming to recover all or part of capital and recurrent costs, recurrent cost recovery being particularly critical to preserve the physical integrity of the system when public funds are not forthcoming. A water charge may also be *an economic tool* designed to conserve water and raise water productivity by promoting: (i) careful management and water conservation; (ii) cultivation of less water-demanding crops and investments in water-saving technologies; and (iii) reallocation of water to high-value agriculture and/or other sectors. Finally, a charge can be *an environmental tool* to counter water pollution and enhance water quality.

Water pricing issues lie at the confluence of two complex 'spheres': on the one hand, the microeconomy of the farm and its linkages to the wider economic system and agricultural policies and, on the other, the hydrology of the plot and its interconnectedness with the irrigation system, the river basin of which it is a part, and the overarching water policy framework (Fig. 2.2).

These nested levels of interaction result in a complex set of dynamics. Economic interactions reflect the multiplicity of factors that govern economic behaviour and the heterogeneity of the different economic actors. Hydrological interactions between upstream and downstream, surface water and groundwater and quantity and quality are compounded by seasonal and interannual variability that creates unstable and unpredictable systems. Economic and hydrological interactions are further embedded within cultural and social contexts that eventually define the distribution of costs and benefits within the society, and are thus highly political in character (Johansson, 2000; Dinar and Saleth, 2005).

In the past, emphasis has typically been placed on influencing the performance of farmers and irrigated agriculture (right sphere) by the manipulation of the

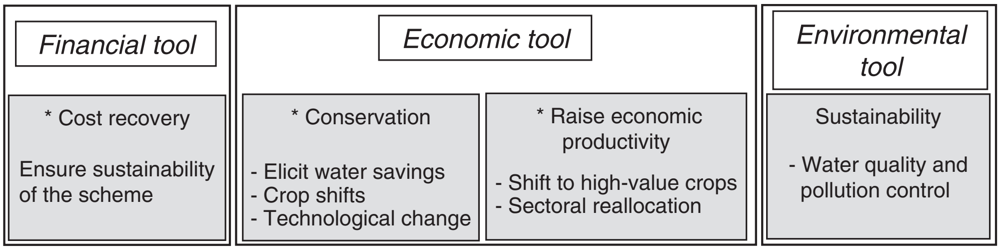

Fig. 2.1. What to charge for?

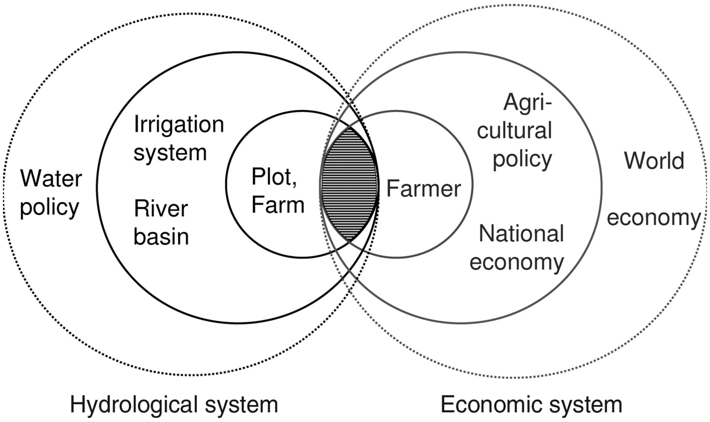

Fig. 2.2. Water pricing issues at the intersection of two spheres of complexity.

hydrologic cycle and the design of canal and pipe networks (left sphere). Increasingly, however, emphasis has shifted to influencing performance of the water system (left sphere) by the adoption of economic and related incentives (right sphere). This chapter reviews the potential and the effectiveness of the latter approach, focusing in particular on the contribution of water pricing. It will argue that water pricing is strongly related to the institutional setting, that is, to the combination of community, government and market regulation, and to the attendant rules that define water governance and management in a particular context. More specialized issues, such as irrigation management transfer, characteristics of water markets, environmental protection, irrigation modernization and politics of water development, though important in their own right and relevant to the issues under consideration, receive less attention in this synthesis chapter, as do related theoretical considerations.

The following section expands on the economic and hydrological systems summarized in Fig. 2.2, and discusses the broad context within which the subsequent discussion is set. Within this framework, we move to examining the practicalities and effectiveness of current water charging practices. The following five sections successively review the main roles commonly attributed to irrigation water pricing: (i) cost recovery; (ii) water conservation; (iii) enhanced water productivity; (iv) intersector reallocation; and (v) control of water quality. The concluding section offers a synthesis of the assessment and corresponding conclusions. While the various sections have been defined for analytical purposes, it will become clear that they are strongly interrelated.

Table 2.1. Evolving priorities of the EU Common Agricultural Policy. (From Gómez *et al*., 2005.)

	Issues and concerns	Objectives	Agricultural water pricing
Past	Poverty in rural areas Increasing food demand	Equity and rural development Food self-sufficiency	Lower prices
Future	Water and soil pollution Budgetary constraints	Sustainable development Economic efficiency	Higher prices

CONTEXT MATTERS

The Economic Context

The rationale for irrigation

For millennia, subsistence and financial self-interest have driven communities to construct village schemes, rulers to develop major projects and farmers to exploit groundwater and make other on-farm investments. During the colonial period, there were those who hoped the self-interest of private investors would drive large-scale irrigation investment, but few such projects proved commercially viable and major irrigation has remained predominantly in the public sector.

Cost recovery has always been a major concern. Communities internalized costs, historic rulers recruited corvée labour mainly from the farming population and colonial governments constantly debated the optimum balance between profitability and income generation. As described by Molle and Berkoff (Chapter 1, this volume), the balance shifted following World War II. Governments and donor agencies continued to pay regard to profitability, re-expressed in economic rather than financial terms (in cost–benefit studies), and also began to raise environmental concerns. But other objectives were often dominant, notably:

- Poverty alleviation, equity and employment generation;
- Regional development and the urban/rural balance;
- Food self-sufficiency and/or food security;
- State building and the search for political support and legitimacy.

These objectives can, of course, be mutually consistent with one another and with economic optimization and environmental sustainability, and such consistency is often claimed. But where they are inconsistent, choices must be made. Despite lip service to economic optimization and sustainable development, large-scale expansion of the irrigated area has, in practice, been driven largely by political interests reflecting these other objectives. Recently, the balance has shifted back in favour of the environment, at least in the USA and Europe, with implications for irrigation water prices (Table 2.1).

Whatever the rationale given for the initial construction of an irrigation scheme, subsequent cost recovery remains a widely accepted policy. In practice, cost recovery is normally limited to the recovery of operation and maintenance (O&M) costs and at most to a (small) share of capital costs. The main driver for cost recovery has been containment of government costs, though recouping at least some of the costs from direct beneficiaries is also advocated on equity grounds. In addition, it is claimed that charging for water can promote favourable economic and financial outcomes, especially if combined with irrigation management autonomy. Some commentators have gone further, arguing that irrigation pricing can lead to economically efficient outcomes. Although such claims are now largely discounted (Molle and Berkoff, Chapter 1, this volume), the idea remains important and is explored later in this chapter.

Cost–benefit analysis

Cost–benefit analysis ostensibly provides the basis for taking decisions on public investments. Standard approaches allow for the adjustment of financial prices as a basis for choosing economically viable projects, with additional studies throwing light on possible economic distortions.[1] The main direct costs are the initial capital costs, which typically account for 80–85% of discounted total costs in surface irrigation. Recurrent costs comprise a higher share in pump schemes though capital costs still largely determine viability. Once built, capital costs are '*sunk*' and the direct marginal costs comprise regular O&M together with the costs of replacement, rehabilitation and modernization. Indirect costs include negative environmental and social externalities and opportunity costs – if any – reflecting an appropriate share of the value of output forgone in alternative uses (see below). The main direct benefits comprise the incremental value of agricultural output *with* relative to that *without* the project. There may also be benefits from domestic supply and other uses, and from positive externalities. If discounted benefits exceed discounted costs, the project is viable.

Although cost–benefit analysis is, in principle, straightforward, its application in irrigation and other water projects has been problematic. Although some claim that ex post evaluation studies show that irrigation projects have performed satisfactorily (Jones, 1995), others suggest that there has been a systematic bias in favour of new construction (Repetto, 1986; Berkoff, 2002; Molle, 2007). Three types of argument support the latter case:

- First, as suggested above, political objectives rather than economic priorities often drive irrigation expansion. Moreover, the political dynamics almost always favour going ahead given the combined self-interest of beneficiary farmers, politicians, contractors, consultants and staff in irrigation, and lending agencies (Repetto, 1986; Merrett, 1997). Finance and other entities serving a broader national interest may restrain irrigation expansion, but can seldom prevent it, even if that is their preference.
- Second, the economic analysis of irrigation is more than usually uncertain. Unwitting optimism is widespread and over-optimistic assumptions are difficult to refute, both with regard to costs and to benefits. 'Costs tend to be high because of: inappropriate design, stemming in part from poor studies done prior to start-up; long gestation periods resulting from funding shortfalls due to changing government priorities and poor capital programming and budgeting; few managerial incentives to control costs; and reported corruption that typically involves kickbacks from construction companies' (Holden and Thobani, 1996). Benefits comprise the difference between two large hypothetical future flows (the values of production *with* and *without* the project). Estimating these flows is based on a host of assumptions that cannot be readily validated (Carruthers and Clarck, 1981; Merrett, 1997; Green, 2003). If prices, yields, irrigation efficiency or cropping patterns are adjusted even modestly, the impact can be surprisingly large. Who is to say the assumptions are wrong?
- Third, the retention of surface irrigation in the public sector and the funding of surface irrigation from the government budget limit financial accountability and help explain why inadequate cost–benefit studies generate such little concern. Canals and related facilities are often classified as infrastructure comparable to roads or power supply, and governments feel responsible for infrastructure. But irrigation is also a productive activity in many ways analogous to industry. Few governments still feel competent to *pick winners* in the industry, yet this is rarely questioned in irrigation.

Cost–benefit analysis is thus malleable, and analysts are invariably under pressure to produce positive results. Feasibility studies that appear competent at the time often prove very over-optimistic in retrospect (Pitman, 2002). Re-estimated rates of return are thus typically much lower at completion of project works than at the feasibility stage, and lower still at impact assessment when actual performance outcomes are available. Moreover, long-term price trends, system deterioration and failure

[1]For example, nominal and effective protection studies.

to account adequately for the *without* case suggest that – even at impact assessment – over-optimism is rife (Berkoff, 2002).

Overriding national priorities

The use of social weights and an opportunity cost for labour are techniques that can, in theory, help address issues of poverty alleviation, equity and employment in cost–benefit analysis (Squire and van der Tak, 1976). These partial equilibrium approaches are, however, controversial, given also the inherent uncertainties described above. Moreover, it is arguable that they do not account adequately for broader issues. Irrigation has both backward and forward linkages, while enhanced incomes have further multiplier impacts. Large-scale irrigation is thus often promoted as the engine that drives rural development as a means to both alleviate poverty and provide job opportunities so as to limit outmigration to cities. Such regional development issues are, in theory, best addressed in a general, rather than a partial, equilibrium context. General equilibrium models are, however, complex and expensive, and well beyond the scope of most project studies. Some advocate a simpler approach, that of increasing benefits by some factor representing multiplier impacts. But, for this to be valid, multiplier benefits should be confined to *incremental* impacts relative to those of the next best alternative, allowing also for opportunity costs and the avoidance of double-counting (Carruthers and Clark, 1981; Gittinger, 1982). It is arguable that such conditions occurred in densely populated Asia at the early stages of development (say, 1950–1980) when other viable regional projects were scarce and labour and water were abundant relative to land. Whether such conditions prevail today, notably in land-abundant Africa and Latin America, is much more questionable. Farmers in these regions often have access to rain-fed lands, population densities are much lower and conventional returns to irrigation have declined drastically.

Even if the case for new irrigation based on multiplier effects is questionable, they may still provide a rationale for *preserving* irrigation that has already been built. If investments in transport, marketing and social infrastructure depend on irrigation for their continued profitability, the case for preserving irrigation as a form of social overhead capital comes into its own (Small, 1990). On the North China Plain, for instance, irrigation is affected by severe water constraints. Water transfers from the Yangtze will help maintain farm incomes and slow rural depopulation. Although new irrigation cannot be justified on economic grounds, the economic returns to the transfer to sustain existing irrigation are strengthened by the costs sunk in existing assets not only in irrigation facilities, but also in rural economic and social infrastructure (Berkoff, 2003a).

Irrespective of these economic arguments, history shows that many schemes have also, in practice, been designed with wider geopolitical motives in mind. The western USA, for instance, illustrates a long history of engagement by the state in support of colonization (Reisner, 1986). The Gezira scheme in Sudan (Gaitskell, 1959), Israeli settlements in Palestine (Lipchin, 2003) and the GAP project in southeastern Anatolia (Harris, 2002) are other well-known examples of projects promoted to achieve geopolitical goals (Molle *et al.*, 2007). Likewise, the context of the Cold War and the food shortages and fears of rural disintegration that followed the El Niño-related climatic perturbation of 1972 did much to justify the huge investments in dams and irrigation infrastructures that were to follow (Barker and Molle, 2004). Food self-sufficiency or food security has often been a top strategic concern to be addressed at any cost. In such situations, economic or hydrologic rationality is in effect neither here nor there and overriding political decisions dictate public investments.

Shifting subsidies and taxation

Moreover, the public subsidies incurred under such rural development policies need to be placed in a general economic context. In the decades after World War II, many countries adopted a policy of taxation of agriculture, notably by export duties (Harris, 1994) and public procurement programmes that maintained farm-gate prices often well below their world price equivalents. The magnitude of this taxation amounted – to borrow from Schiff and Valdés (1992) – to a 'plunder' of agriculture during 1960–1985. In Mexico, the price distortion amounted to an implicit tax of 20–50% of the value of the project commodities (Duane, 1986) and similar state extractive policies were carried out in most developing countries, including Egypt (Barakat, 2002), Thailand (Molle, Chapter 5, this volume), Malaysia (World Bank, 1986), Pakistan (Chaudhry *et al*., 1993), Côte d'Ivoire, Ghana and Sri Lanka (Krueger *et al*., 1991; Schiff and Valdés, 1992). Low food prices benefited the urban poor and landless, and taxes on output generated public savings for investment in industrial and urban development, only partially offset by irrigation and other rural subsidies (Lipton, 1977). Low food prices also had adverse impacts on crop output so that rationing was often required to manage consumption, limit imports and maintain food self-sufficiency.

Over time, the arithmetic of relative taxes and subsidies changed drastically as world prices declined and incomes rose. This and the widespread adoption of liberalization policies led to the abolition of most export duties and food-rationing programmes. Reforms initially boosted farm output and incomes as farmers responded to liberalized markets and exploited the agricultural technologies open to them. But as prices declined further, and as economic growth and diversification took place, urban/rural income differentials were reaccentuated, often provoking farmer unrest. Fearing also adverse impacts on domestic output,[2] some governments (e.g. China and India) have begun to support (rather than – as in the past – tax) farmers by limiting imports and adopting other trade-distorting measures. In this they have followed the lead of developed countries (the EU, the USA and Japan) that have long protected agriculture. This situation helps explain the reluctance of governments to raise water charges or other input prices for fear of losing their competitive edge (Tiwari and Dinar, 2001), since many farmers have to compete with exporters from the North who benefit from lavish subsidies.[3]

These trade distortions (market access, tariffs and export subsidies) are the major concern of the WTO Agricultural Agreement (WTO, 2000). Their removal would raise farm-gate prices significantly by reducing developed country exports, thus moderating the need for interventions by developing country governments in support of their farmers, besides facilitating attainment of food self-sufficiency objectives and promoting developing country food exports and inter-south trade (USDA, 2001). The WTO agreement also aims to reduce direct food and fertilizer as well as other input subsidies that have a direct impact on trade. In contrast, irrigation expenditures are amongst those that can be used freely since it is argued that they have minimal impact on trade (WTO, 2000). This is perhaps debatable. It is true that viable irrigation projects do not distort trade but if – as suggested above – much irrigation has been uneconomic, cumulative worldwide irrigation subsidies have contributed to declining world prices in a manner comparable to that of other trade distortions. Moreover, although irrigated output has risen enormously, rain-

[2]Taxation of agriculture and the resulting 'urban bias' are also seen as reflecting the shifting influence and political clout of interest groups and coalitions (whether defined by sector or income groupings) (Lipton, 1977; Bates, 1981; Sarker *et al*., 1993), linked to their income, information and education, potential for collective action and political representation (Binswanger and Deininger, 1997). According to Bates (1993) this transformed the agriculture sector from 'an embattled majority that is taxed into a minority powerful enough to be subsidised'.

[3]Yang *et al*. (2003) show how decreasing profitability could put further pressure on domestic food production in China, challenged by international markets since the late 1990s, and even more since China's recent accession to the World Trade Organization (WTO) (Huang and Rozelle, 2002). After adhesion to the WTO, Jordan had to face 'unfair market intrusions by countries with less stringent WTO membership conditions' (WTO, 2001) and realized that abolishing subsidies altogether would be detrimental to its own farmers.

fed yields and output may well have been suppressed (Berkoff, 2003b). If so, food self-sufficiency based on irrigation may have been achieved at the expense of the rain-fed farmer.

Ultimately, all tax and subsidy polices are conditioned by politics, and reflect the cultural, economic and political milieu in each country concerned. Although the WTO negotiations aim to moderate economic distortions, and thus benefit those that are discriminated against, especially by developed country interventions, all such interventions must be understood within the wider political and policy context if they are to be analysed and possibly changed (Sampath, 1992; Speck and Strosser, 2000).

The Hydrological Context

The characteristics of water and water use

The physical characteristics of surface water are well known and include site-specificity, mobility, stochastic variability and uncertainty, bulkiness and solvent properties. Accompanying these are its relatively low value as a commodity, the economies of scale that often make supply a natural monopoly and the pervasive interdependence of water users (Young, 1986; Livingston, 1995; Morris, 1996; Savenije, 2001; Green, 2003). Groundwater shares some of these attributes but has other attributes that set it apart, including its relative immobility, security and divisibility.

Water has numerous human uses, some of which are consumptive (agriculture, industry and domestic) and others non-consumptive (fisheries hydropower, navigation, etc.). Water also has environmental values that are appreciated by humanity. The characteristics of water use in agriculture set it apart in many ways from its use in municipal and industrial use.

Diversions for consumptive use are invariably larger than the fraction that is actually consumed, with the balance returning to the water system. Agricultural withdrawals (predominantly for irrigation) account worldwide for 70% of the water withdrawn for consumptive use (Aquastat, 2004). Its share is typically higher in developing than in developed countries. Evapotranspiration accounts for 40–60% of agricultural withdrawals (rising to above 70% due to repeated reuse, modern irrigation techniques, etc.). In contrast, domestic water withdrawals are largely used for washing and cooking, and domestic diversions largely return – often in a polluted form – to the water system. Similarly, industrial diversions are mainly for cooling and dilution of wastes rather than for chemical incorporation in products. Consumptive use as a proportion of withdrawals is thus much higher in agriculture (70%) than in domestic (14%) or industrial (11%) use, and agriculture accounts for as much as 85–90% of total consumptive use worldwide (Shiklomanov, 2000).

Uses in the municipal and industrial (M&I) as well as the irrigation sectors are not always fully interchangeable. M&I use is usually far more valuable than in irrigation, and logic implies that water should move wherever possible from irrigation to M&I in the event of conflict. But transfers are only feasible if the infrastructure is, or can be, integrated at acceptable cost. Moreover, M&I have much higher quality and security-of-supply requirements than irrigation, which may limit transfer opportunities.

Consumptive use impacts on non-consumptive uses through its effect on flow regimes, water quality and flood risk. Given that irrigation use is so much greater than M&I use, the major quantity conflicts are generally between irrigation on the one hand and in-stream and environmental uses on the other (though M&I can have large quality impacts). Irrigation diversion capacity often exceeds dry season flows and, as use rises, irrigation may be able to divert flows year-round. In-stream uses suffer, rivers and wetlands dry up, affordable groundwater is exhausted and pollution loads rise (though flood risks may moderate). Action to safeguard in-stream and environmental uses may then become desirable and, in effect, irrigation rather than the environment becomes the *user of last resort* (Elston, 1999).

Irrigation efficiency

The concept of irrigation efficiency is often misstated (Willardson *et al.*, 1994; Frederiksen, 1996; Keller *et al.*, 1996; Huffaker *et al.*, 1998; Perry, 1999; Huffaker and Whittlesey, 2000; Loeve *et al.*, 2004; Molle and Turral, 2004) with significant implications for water pricing. If water is abundant, scheme-level efficiency is of limited concern other than for system capacity and capital cost reasons. If basin water is scarce, raising scheme efficiency can be elusive since return flows are fully utilized and the only *additional* source of water lies in reducing unproductive losses.[4] In north China, for instance, apart from uncontrollable floods and releases for silt and pollution control, little water reaches the sea from a vast area containing up to 7.5% of world population. Drainage and wastewater reuse are pervasive, losses recharge groundwater, farmers underirrigate, tail-end areas are abandoned and basin efficiency is high by any standards. Existing irrigation can essentially absorb *all* the water available and shortages relative to theoretical crop water requirements have little meaning (Berkoff, 2003b).

It is not only basin efficiency that is misstated. Scheme and on-farm efficiencies are also often (much) higher than assumed. That water is 'wasted' when it is abundant (e.g. after it rains) is inconsequential – low physical efficiency may even correspond to high economic efficiency since management is eased and labour reduced (Gaffney, 1997). In contrast, farmers fight for water and return flows if it is scarce (and overpump groundwater). The struggle for water *when it is scarce* means that little water is wasted *when it has value* and average estimates of efficiency can be very misleading. Case studies from Thailand (Molle, 2004), California (Zilberman *et al.*, 1992) and China (Loeve *et al.*, 2003) have shown the multifarious efforts deployed by farmers to adjust to water scarcity and make the best use of water. These changes go often unnoticed but statements such as 'farmers waste water just because [they] are not aware of the fact that water has a value' (Roth, 2001) are both unfair and mistaken. Moreover, even if there is potential for increased scheme-level and on-farm efficiency, this can require expensive investments in drip or sprinkler systems that may not be justified either financially or economically.

Irrigation design

Opinions on irrigation design range from those that advocate modern systems of control (Plusquellec, 2002) to those that advocate simple technologies that respond to human and institutional limitations (Horst, 1998; Albinson and Perry, 2002). The critical factor is stochastic water variability: from day to day, week to week and year to year. *Supply* is variable because runoff is variable; *demand* is variable because rainfall and crop water requirements are variable. Reservoirs and groundwater improve predictability, and on-demand systems help farmers obtain water when it is needed. But in practice, most surface water systems are designed to meet peak water requirements for a specified cropping pattern, say, 3 years in 4 (i.e. the 75% year) (the full area being irrigated in the wet season and a restricted area in the dry season). This is a compromise. If greater security is guaranteed to a smaller area, in most years the available resource is underutilized. If canal capacity is increased to expand the area in good years, unit costs rise, security declines and capacity in most years

[4] That there is little water – if any – to be saved in closed basins must, however, be qualified since there are notable exceptions. If return flows from irrigation are degraded in terms of quality (salinity, contamination), they may incur yield losses when reused (Morocco: see Hellegers *et al.*, Chapter 11, this volume; Pakistan) or be unfit for agriculture (e.g. Jordan Valley: Fontenelle *et al.,* Chapter 7, this volume), and therefore losses should be minimized. If the time taken by water to become available again is very long (e.g. percolation to deep aquifers), these volumes are not available for short-term use. Water wasted in the wet season in cities or irrigation schemes could also sometimes be kept in reservoirs for later use in the dry season. Another caveat concerns the costs incurred by possible successive pumping operations associated with reuse.

is excessive. In contrast to fully on-demand systems, therefore, it is *by design* that the full area cannot be irrigated in dry periods, in dry years and during the dry season.

As economies develop, shortages increase, water tables fall, other users get priority and variability is increasingly concentrated on irrigation as the residual user. Both the value of water and the costs of insecurity rise. Reservoirs are built, farmers install wells and on-farm ponds and modernization and volumetric measurement become affordable. Operator salaries and skills also rise in line with general living standards. In other words, irrigation responds to the external context. Ultimately, the issue in irrigation design is not that it is innately different to M&I design, but that there is a continuum from simple surface systems suited to low-return agriculture in poor countries, through conjunctive use and partially modernized systems appropriate to countries moving through the rural transition, to advanced technologies appropriate to high-return agriculture in richer countries that are completing the transition. At the limit, design approximates to that for M&I, and volumetric measurement at the level of the individual farmer becomes feasible. Until this point is reached, physical characteristics of irrigation severely constrain the possibility of using efficiency (marginal cost) pricing, and the debate on how economic pricing can be introduced has, in general, been a distraction.

Irrigation performance

Irrigation performance also ranges through a continuum. Traditional systems can be stable, but crop yields and farm incomes often remain low. Productivity and income in public systems are normally higher and manageability improves as an economy develops, agriculture becomes more entrepreneurial and market-driven, farm sizes and incomes rise, O&M agencies are better-funded and accountable and storage and modern control become affordable, manageable and justified.

Nevertheless, despite these trends, in the view of most observers, irrigation performance in developing countries remains generally poor. Water variability is again the main reason why so many schemes are so difficult to manage. *Ex post*, management must respond to conditions that deviate continuously from the average conditions implied by a design cropping pattern that means little to the farmer. Irrespective of design intentions, the farmer typically *wants* more water than he is allowed in the dry season and in dry periods; after rainfall, he may reject his allocation even if this causes problems elsewhere in the system. Differing objectives set up a continuing tension between scheme managers and farmers. Farmers interfere in outlets and water levels contributing to head-end and tail-end problems, while poorly paid system operators living close to the farmers fail to enforce – perhaps cannot or do not want to enforce – the rules. On the one hand, water-use efficiency is enhanced as farmers struggle for water and, on the other, damage is pervasive, inequities emerge and there is a broad failure to operate the system in line with design.

A Typology of Irrigation Systems

Figure 2.3 suggests a simplified typology of irrigation systems that reflects the above discussion. It classifies systems in relation to an index of relative water supply (RWS)[5] and suggests two broad types of management response: *pragmatic management* and *volumetric management* (that are linked not only to the degree of development, but also to the climatic context). With respect to Fig. 2.3:

- Situation W1 is typical of wet regions with abundant water supply. Water tends to be supplied continuously – often for paddy – at, or close to, full supply level, though rotations can be necessary if main canal capacity is a constraint. Occasional shortages may occur due to ill-discipline and farmer

[5]RWS is defined as the ratio of the water delivered to gross irrigation requirements (net of the effective rainfall) after accounting for losses. It provides a broad indication of the amount supplied relative to demand.

intervention. Minimal data on flow, rainfall and land use are typically collected.

- Situation W0 typifies non-arid countries as water is increasingly exploited. Operations reflect experience rather than active management, with water often released in response to farmers' complaints. Head-end and tail-end problems are limited while temporary supply reductions can lead to short-term crises as discipline breaks down. Data are collected haphazardly and seldom analysed. As RWS falls to 1, conflicts intensify and rotations are increasingly adopted.
- As RWS drops below 1 (D0), rotation becomes the rule. Farmers respond by deficit irrigation and conjunctive use (tapping drains, ponds or aquifers) and use water more carefully. Head-end and tail-end problems become pervasive. Data are collected more systematically and basic parameters (efficiency and water applied) are calculated. Supply-driven management predominates with scheduling planned, based on target allotments, and bulk allocations may be negotiated.
- Under situation D1, potential demand cannot be met and supply limits allocations. If the system is uncontrolled, water distribution may be chaotic. Groundwater replaces surface water and conjunctive use is ubiquitous, with land left fallow or abandoned. In systems that are better controlled – depending on design – water is confined to part of the scheme, supplied in turn or allocated proportionally (as under warabandi).[6] In fully controlled systems, volumetric rights are clearly defined and water may be supplied on demand, subject to availability.

When RWS falls below 1, the crucial step is the shift from 'pragmatic' to 'volumetric' management (Fig. 2.3). *Pragmatic management* is weak, reactive and ad hoc, with managers responding to complaints from below and farmers responding as best they can, e.g. by investing in wells and on-farm storage. As scarcity develops, water distri-

[6]All systems have to cope with hydrological variability (i.e. varying values of RWS) but both demand and supply are more predictable in arid climates since rainfall is a less significant factor and reservoirs are the norm. In humid climates, rainfall is a much more complicating factor since it strongly influences not only supplies at the source, but also requirements in the fields.

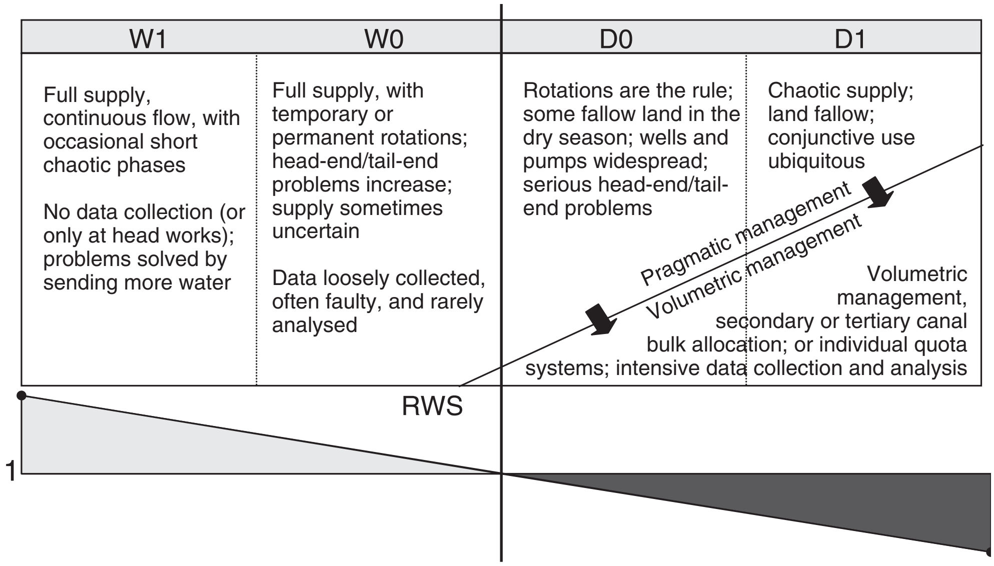

Fig. 2.3. A typology of irrigation systems.

bution becomes increasingly chaotic. Such conditions are common in developing countries, especially when schemes are large, farmers are numerous and poor and surface irrigation is dominated by cereals and low-return crops. Under these conditions, head-enders tend to divert what they want and tail-enders often fail to obtain even minimal supplies. With *volumetric management*, in contrast, a stronger degree of control is maintained. Water may be allocated in bulk or by individual quotas, rotational rules are clear and roughly predictable and risks are defined. At the limit, water may be provided approaching on-demand supply. This situation tends to occur in developed and/or arid countries, especially when farms are large, irrigated agriculture is for high-return crops and farmers incur large on-farm costs and financial risks (see above). Security in supply invites complementary on-farm investments and tends to make farmers willing to pay for water since even high charges comprise a small share of farm costs and service standards are critical.

This classification simplifies real-world diversity and variability. Even so, it can provide guidance in assessing the potential of water pricing policies. The difference between pragmatic and volumetric management corresponds to a 'quantum leap', and efficiency pricing is only possible if the scheme is under volumetric management and control is maintained. Many reforms fail because they assume very lightly that shifting from the former to the latter is simply a question of goodwill or capacity building, whereas it is linked in complex ways not only to RWS, but also to irrigation design and hydraulic control, manager-incentive and farmer-incentive structures and the wider institutional context.

Implications for Irrigation Pricing

Full marginal cost pricing

By analogy with domestic water supply and other infrastructural services, some analysts recommend long-run marginal cost (LRMC) pricing in irrigation (Arriens *et al.*, 1996). But there are important differences between the sectors. One issue is that volumetric pricing is far more problematic in irrigation than in reticulated urban systems, and this greatly restricts the adoption of efficiency pricing in irrigation. Basically, LRMC pricing in the urban sector simulates a competitive market price for a final good and, besides funding recurrent, replacement and related costs, it aims to generate the investment funds needed to match rising demand as a city expands and its population becomes richer (Munasinghe, 1990). If consumers are willing to pay the LRMC price, system expansion is economically justified; if not, effective demand can be met by existing capacity.

In contrast, irrigation water is an intermediate, not a final, good, and canals are sized to serve a specific command area at defined levels of probability (see earlier section). Possibilities for system expansion are thus restricted. Since charging existing farmers for a new scheme is no more justified than charging City A's inhabitants for expansion of City B's system, initial capital costs should usually be treated as *sunk*, in which case marginal direct costs comprise O&M and replacement costs.[7] Of course, if the scheme is inherently profitable, farmers should, in theory, be able to repay full costs (including initial capital costs), and charging them less than full cost gives them a windfall gain. But if expansion of irrigation has been driven by other public objectives (see above) and is uneconomic, charging full capital costs is neither feasible nor equitable (Carruthers and Clarck, 1981). Moreover, over time, capital subsidies are incorporated in land values and, though the initial beneficiaries may receive a windfall gain, inequities arise if charges are imposed on those that subsequently buy irrigated land.

Irrespective of any theoretical rationale for marginal cost pricing, there may still be a case for charging farmers a share of initial capital costs on financial and equity grounds, given

[7] They should also, in theory, cover modernization and system expansion costs if the water saved by the modernization investments is justified specifically in terms of the expansion of the scheme. The analogy with LRMC in expanding urban systems is then valid.

the needs of the economy and adverse impacts on rain-fed farmers. There is also the quite separate issue of whether opportunity values in alternative uses and externality costs should be reflected in some way in the irrigation charge. But competition between irrigation and cities is limited to specific periods and locations and, once urban demands are satisfied, opportunity cost falls drastically. Beyond compensating farmers on a case-by-case basis, water pricing to promote reallocation is generally impracticable (Molle and Berkoff, 2006; more on this later). Once M&I use is met, most conflicts lie between irrigation and the environment. But valuing environmental *externalities* (third-party impacts, soil salinization, water contamination, health hazards) is also a contentious issue, and willingness-to-pay for moderating such costs varies greatly at differing locations and stages of development. In most cases, there is no agreement on how pricing can mitigate negative impacts, and reflecting environmental use and valuing externalities are again impracticable (see section *Pricing as an environmental tool*).

The relevance of marginal cost pricing

Moreover, the need for strict marginal cost (efficiency) pricing in practice is often questionable. As argued above, irrigation performance typically reflects a rational response by farmers and operators to the evolving context and associated incentives. Water is used much more efficiently than is commonly supposed, and the scope for enhanced water-use efficiency and the potential role of water pricing can be greatly overstated. Furthermore, the massive expansion of private groundwater, much of it within surface schemes, has further strengthened irrigation performance. Groundwater is, in effect, available on demand and provides a security of supply that can offset variability of rainfall and canal supplies. Groundwater use, or conjunctive management, has thus accounted for most of the high-return diversified agriculture that has developed in response to economic growth, urbanization and external markets, and groundwater's pervasiveness limits the need for surface irrigation to meet these diversified demands.

In addition, no administered price can reflect short-term stochastic variability and, though at the margin water charges may impact on farmer behaviour and promote favourable economic and financial outcomes (Fig. 2.1), this is far short of true economic efficiency pricing. Modern control systems may be justified and, at the limit, a pressurized on-demand irrigation system approximates to a reticulated urban network. But, while urban systems are, in principle, *designed* to operate on demand, the vast majority of surface irrigation projects *by design* cannot supply water on demand since they cannot meet potential farmer uses when water is scarce (e.g. in the dry season or a drought). Comparing benefits and costs at the margin is therefore meaningless because farmers cannot, like urban users, access as much water as they wish and are willing to pay for. These considerations suggest that efficiency pricing is usually impracticable even in fully reticulated systems; supply management and rationing will inevitably remain the preferred mechanisms for controlling surface distribution in most irrigation in developing countries.

Potential price effects

As empirical evidence will confirm, the economic and hydrological characteristics reviewed above impact on irrigation water pricing in such a way that water charges are eventually, first and foremost, a cost-recovery mechanism. Even confining water charges to this one objective is far from straightforward since, as discussed above, what is meant by cost can vary depending on whether costs are limited to financial costs or extend to the full economic costs to society (Rogers *et al.*, 1998) and what is to be recovered may be limited to recurrent and replacement costs or include some or all of the capital costs invested. Financial O&M costs are invariably a priority since, once a scheme is constructed, production is contingent on continued O&M of the infrastructure.

In addition to financial cost recovery, economists argue that opportunity and

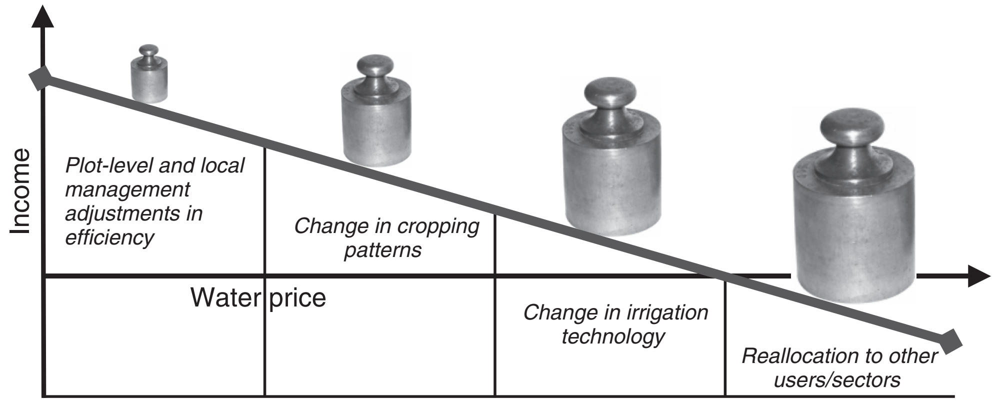

Fig. 2.4. Effects of water pricing as an economic tool.

externality costs are equally valid in societal terms (Rogers *et al.*, 1998; Tsur, 2004). Although their definition and estimation vary, the level of water charges may impact on farmer behaviour and bring economic benefits. Figure 2.4 proposes a tentative hierarchy of responses to increasing water prices, while recognizing that the order of these effects may sometimes be altered by relative factor prices and other aspects. Moderate water prices may trigger low-cost adjustments in water management, while higher prices may successively elicit changes in cropping patterns, in irrigation technology and, finally, release water to other higher-value activities. These effects imply a role for pricing as an economic tool and the likelihood of achieving such outcomes is examined in the following sections.

A Note on Terminology

A *water charge* can be defined as an actual (financial) payment by users to access water and is the term generally adopted in this chapter. It is equivalent to a *tariff*, a term commonly used in the domestic sector when differential rates are set. *Charge* is a term disliked by some decision makers, who fear that it suggests that water – perceived as a gift of nature or god – is taxed. In 1979, several Asian countries agreed to replace it with the term *irrigation service fee* (ISF) (ADB, 1986a). This is now often adopted, though it conflicts with the definition of a *fee* as an administrative payment (e.g. for the registration of a water right). Another term commonly used is *water price*. This is preferably confined to the (economic) price that emerges in a market as the result of the actions of willing buyers and willing sellers, with no connotation of (financial) cost recovery. Since such markets are rare in the water sector, *price* is often used as a synonym for *charge* to indicate the administrative rate set by an agency to a user. Most of the discussion in this chapter uses the term *water charge*, focusing on how water charges are reasoned, justified, determined, enforced, recovered and eventually expended.

A word is also necessary on the terms *ability-to-pay* and *willingness-to-pay*. Many studies conclude that farmers have an *ability-to-pay* much higher water charges than are charged in practice. This is sometimes supported by evidence that they are *willing-to-pay* much higher amounts for private irrigation and by the fact that consumers in the domestic sector are *willing-to-pay* much higher prices to street vendors than the tariffs charged by the utility. The use of these terms can, however, be confusing.

Willingness-to-pay is best used as an economic term to describe consumer behaviour. The poor may be *willing-to-pay* the high unit price charged by a private tube well owner or a vendor but buy little at this price, the amount being determined by profitability (in irrigation) or subsistence needs (in domestic use). As prices and incomes shift, demand also shifts reflecting the price, income and cross-price elasticities described by standard demand curves (Young, 1996). Similarly, private investments (such as wells) and their subsequent operation reflect investor assessment of profitability, that is, by farmers' *willingness-to-pay* (*or to invest*). Purchases from a private tube well owner or vendor and private investment in irrigation are determined in markets governed by the actions of willing buyers and willing sellers.[8]

If *willingness-to-pay* describes behaviour, *ability-to-pay* relates to farmer incomes and public subsidies. If irrigation investment is economically justified, and prices are undistorted, farmers should in principle be *willing-to-pay* all costs including capital cost. But irrigation is driven by non-economic objectives and in most cases farmers *should not* repay full capital costs. If they are unable to pay for marginal (future) costs, then – leaving aside distortions in other costs and prices – continued irrigation is itself uneconomic. In extreme cases, farmers may be unable to pay even recurrent costs since the resulting farm incomes are inadequate to sustain life (Cornish *et al.*, 2004) or the rainfed option is more profitable. But the issue in irrigation is seldom, if ever, an absolute inability-to-pay (although this may, of course, typify extreme cases in respect of domestic water). It is one of fairness, incentive and acceptability, and *ability-to-pay* is best thought of as that level of payment thought reasonable and practical, given the general context and government priorities and objectives. The level of subsidies given to construct a new scheme or sustain an existing scheme is thus ultimately a political decision.

[8]Such markets may, of course, be distorted as a result of monopoly practices, distorted input and output prices, changeable public policies, etc., and there may be a case for interventions by government or a regulator to correct for these distortions. They are also shaped by social relationships and values.

CHARGING FOR WATER IN PRACTICE

This section addresses the practicalities and modes of charging for water, as well as the current situation regarding cost recovery by irrigation schemes.

Main Types of Water Charge

The following are the most common ways of defining charges and their differentiation according to uses and users (Sampath, 1992; Tsur and Dinar, 1997; Garrido, 1999; Bosworth *et al.*, 2002; Easter and Liu, 2005):

1. *Uniform user charge* – users are taken to have similar access and are charged evenly. Even if the level of use varies, differences cannot, or are too costly to, be assessed.
2. *Area-based charge* – the irrigator is charged according to the area irrigated, based either on: (i) the area owned; or (ii) the area cropped (declared by the farmer or assessed by the agency).
3. *Crop-based charge* – the charge is based on area *and* type of crop. Differentials may be justified by crop priority (e.g. cereals for food security) or water diverted or consumed by crop or its value.
4. *Volumetric charge* – water is charged, based on actual diversions to a user or group of users (bulk water pricing). Metering is necessary but volume may be represented by time or the number of 'turns', provided discharges are more or less stable and predictable.
5. *Volumetric block tariffs* – when metered, charges can be fixed for different levels of consumption. Increasing block tariffs discourages excessive use. Decreasing block tariffs promotes sales and rewards economies of scale, being appropriate only if water is abundant.
6. *Mixed tariffs* – charges combine a flat rate (usually area-based) with a volumetric

charge. This provides both a stable minimum revenue to the operator and a variable charge according to use.

7. *Quotas at fixed charges* – quotas may be uniform (e.g. based on area) or vary by crop. Charges can be proportional to nominal volumes or vary with crop type (as in the Jordan valley).

8. *Quotas and marginal volumetric pricing* – users can access more than their quota (subject to availability and within limits), but additional use is charged at higher rates (as in Israel).

9. *Market-based price* – the price of water is determined in a market where allotments can be traded (within season, seasonally or permanently). If the market is regulated, the regulator may set the price, set price limits, serve as broker, etc. (as in the California Drought Bank).

Each method has its advantages and disadvantages, notably the ease with which charges can be calculated, justified and implemented. Additional modalities may also vary: for instance, charges may vary by season, be paid before or after cropping, in one or more instalments, in cash or in kind, etc.

Besides direct charges, farmers may also be charged implicitly via the tax system or in the level of output prices. Land taxes, for instance, often vary to reflect the higher productivity of irrigated land, and betterment levies may be imposed when irrigation is brought to an area for the first time. Similarly, procurement programmes and/or export duties can depress crop prices and can be thought of as an indirect charge. But this is not specific to irrigation and may be offset by other subsidies (e.g. on fertilizer). Moreover, farmers may be protected rather than taxed. These and related issues are thus best considered in relation to the general context rather than to irrigation charges per se (see earlier section).

Who Collects and Uses the Water Charge?

Water charges may be assessed and collected by the state, by a revenue or irrigation department, or by a combination of the two (as in much of India); by an autonomous irrigation entity at the national level (as in the case of the National Irrigation Administration (NIA) in the Philippines) or at the scheme level (as in China and other countries where schemes are managed autonomously or quasi-autonomously); or by a communal organization (such as a Water User Organization) collecting charges directly from its members. Numerous options exist. The state may assess and collect charges at farm level, and consider this levy as revenue. Alternatively, assessment and spending of this revenue can be shared with other levels. Again, a Water User Association (WUA) or some other agent may collect the fees and retain a pre-assigned share for its own requirements (e.g. O&M of the tertiary command), transferring the balance to the irrigation agency, the basin agency or the state, in return for irrigation supply. This can be paralleled by contractual arrangements made for bulk allocations and schedules at each level (e.g. between the river basin agency and irrigation entities, between the irrigation entity and pump/canal organizations and between the canal organization and the WUAs).

In other cases, a state or provincial government may regulate the different rates applied by various entities (including the charge paid by farmers), or each entity or organization may be free to establish its own rates subject to agreement between the different levels and approval under the rules of the organization. Where the state is responsible, payment may be reduced or forgiven in a drought or for some other reason.

There are also options relating to incentives and farmers' involvement in decision making. For instance, incentives may be provided to encourage collection either being paid to officials of the relevant organizations or to private subcontractors. The corresponding levels of farmers' involvement in decision making are equally important (e.g. in allocation decisions or possibility of hiring their own staff). The nature of the arrangements impacts on the rate of collection and on the potential for water conservation and enhanced water productivity, as discussed further below in the appropriate sections.

Who Pays What and How Much?

Types of charge

The most common form is area-based or area plus crop-based, as in Pakistan (Bazza and Ahmad, 2002), Nigeria (Olubode-Awosola *et al.*, 2006), Kazakhstan (Burger, 1998), Vietnam (Fontenelle *et al.*, Chapter 7, this volume), Turkey (Yercan, 2003), Argentina, Greece, Japan, Philippines and Sudan (Cornish *et al.*, 2004), with occasional distinctions by season (as in India, Saleth, 1997; or Nepal). This type of charge accounted for 60% of the sample studied by Bos and Wolters (1990).

Volumetric pricing is usual in the Middle East or North Africa, e.g. Tunisia (Hamdane, 2002a), Iran (Perry, 2001a,b), Jordan (Venot *et al.*, Chapter 10, this volume) and in countries such as the USA, Australia, Southern Europe and Mexico. Volumetric pricing is often associated with a quota, and defined at a bulk rather than at an individual level. Two-part tariffs are also common (e.g. Spain: Maestu, 2001; Colombia: Garcés-Restrepo, 2001; Lebanon: Richard, 2001; Morocco: Ait Kadi, 2002). Volumetric charges are widespread in lift irrigation given the ease of measurement (though not in Vietnam; see Fontenelle *et al.*, Chapter 7, this volume).

Numerous variations occur: in Indonesia charges may be differentiated by head, middle and tail, and be lower in unproductive areas (Hussain and Wijerathna, 2004), and in India they sometimes reflect water dependability (Sur and Umali-Deininger, 2003). In Bangladesh, at one time charges were set as 3% of gross incremental benefit but this proved impracticable (ADB, 1986b). In contrast, simpler approaches may be negated by considerations of equity: a flat per acre rate was, for instance, adopted in Sind in 1972 to reduce irregularities only to be abolished in 1980 since charges based on actual crop areas were thought fairer. Some countries once collected charges in kind (e.g. the Office du Niger, Mali: Aw and Diemer, 2005; Philippines: Oorthuizen, 2003), and in Tanzania this is still an option (Tarimo *et al.*, 1998). Elsewhere, rates are expressed in terms of a paddy quantity (e.g. in Vietnam and Philippines), though rates must be updated if productivity or prices vary (Carruthers *et al.*, 1985).

Some countries impose a resource charge in addition to an irrigation charge. This may simply be an administrative fee, e.g. for registering a water right, but can be a contribution to basin management costs South Africa (Spain, France: Berbel, Chapter 13, this volume; Tanzania: van Koppen *et al.*, Chapter 6, this volume; Colombia: Garcés-Restrepo, 2001). Resource charges are seldom significant to the farmer (e.g. 13% of O&M costs in Peru: Vos, 2002).

Despite occasional claims that models can assist in determining technically optimal prices (Tarimo *et al.*, 1998; Louw and Kassier, 2002; Garrido, 2005), there is little evidence that this has ever occurred: charges are invariably based on historical practice, microeconomic data on crop income or the level of O&M/investment costs (Lee, 2000) and are the result of negotiations or bureaucratic arbitration (Lanna, 2003). In general, a balance is struck between supply costs and what farmers can pay or, maybe more to the point, between tax collection costs and higher charges that would not be politically possible.

Charging mechanisms are not necessarily established once and for all and may evolve with circumstances and objectives (Rieu, 2005). Changes may be triggered by climatic circumstances (volumetric pricing will perform badly in dry years, as experienced in Mexico: Kloezen, 2002), level of state subsidies, O&M costs (which may vary with age of the system), type of incentives needed, etc. (see Plantey *et al.*, 1996; Nicol, 2001 for two French examples).

Rates of recovery

Collection problems have plagued many systems (World Bank, 2005c). Collection is low in Pakistan (30–60%: Bazza and Ahmad, 2002; less than 30% in Sindh: Cornish *et al.*, 2004; and 5–15% in schemes studied by Hussain and Wijerathna, 2004), Kenya (20% in West Kano: Onjala, 2001), Nepal (5%: World Bank, 1997), Bangladesh (less than 10%: World Bank, 2005c) and India (8% in 1989: Saleth, 1997), though 66% and 85% in Andhra Pradesh and Uttar Pradesh, respectively, in 1998 (Sur and Umali-Deininger, 2003).

Recovery rates tend to be higher: (i) under authoritarian governments; (ii) if supply is cut off for non-payment; (iii) if charges are low, recovered with other taxes and/or collected before the crop season; (iv) where users decide on the use of the charges; and (v) when supply is reliable. Thus, it is 98% in Mali (Office du Niger: Aw and Diemer, 2005), 95% in Turkey (Özlü, 2004), 90% in Syria (Bazza and Ahmad, 2002) and Tunisia (Hamdane, 2002a), 80% in Mexico (OECD, 2003) and the Jordan Valley (Venot *et al.*, Chapter 10, this volume) and 50% in Kyrgyzstan (Sehring, 2005). The overall rate of recovery for a sample of 82 irrigation providers was 77% (Lee, 2000).

Water charges come with both administrative and compliance costs that can be quite substantial (Nickum, 1998; Tiwari and Dinar, 2001; Johansson *et al.*, 2002) and differ depending on the type of charge (Tsur and Dinar, 1997). In Bihar, collection costs are said to sometimes exceed the income derived, being estimated at between 52% and 117% of the amount collected (Prasad and Rao, 1991). For Bhatia (1991), collection keeps '5,000 persons busy and unproductive in the fields'. Transaction costs make volumetric charging impractical in Egypt (Bowen and Young, 1986) and similar settings.

The burden of irrigation charges

This burden varies widely. Bos and Wolters (1990) reviewed 150 systems and, in all but one, water charges were less than 10% of the net farm income excluding water costs. The share ranges from zero if water is supplied free (as in Albania, Poland, Croatia: Cornish *et al.*, 2004, Saudi Arabia: Ahmad, 2000, Thailand: Molle, Chapter 5, this volume and Taiwan) to above 30% in pump schemes (e.g. 31% in Niger: Abernethy *et al.*, 2000; 34% in Gujarat: Cornish *et al.*, 2004; and even 65–76% in the Jordan highlands: Venot *et al.*, Chapter 10, this volume). Figure 2.5 shows the ratio for a number of schemes and scheme averages.

Two qualifications should be added here. First, formal charges do not capture in

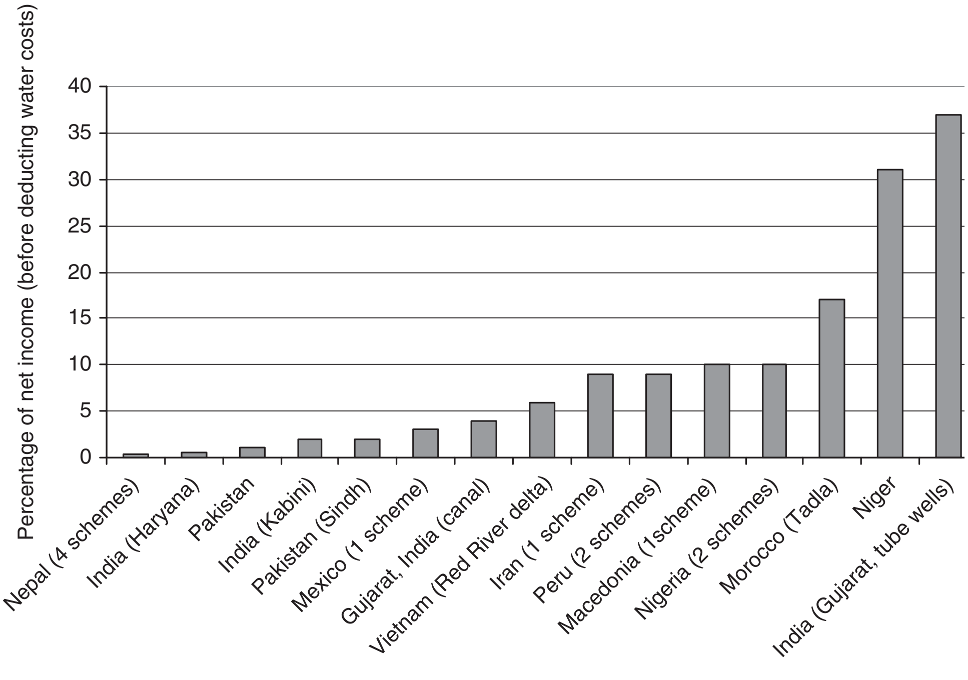

Fig. 2.5. Water costs as percentage of net income.

full the water payments made by farmers. Extralegal payments to local officials are widespread, especially if water is scarce (India: Wade, 1982; Indonesia: Rodgers and Hellegers, 2005; Vietnam: Fontenelle *et al.*, Chapter 7, this volume; Pakistan: Rinaudo, 2002). Farmers are also usually responsible for O&M costs within the tertiary – watercourse – command (in Egypt, India, Pakistan, Indonesia, etc.). Finally, farmers incur major on-farm costs including investments made to augment and/or offset insecurity in main system supplies (not only in private tube wells, but also in hand pumps, reuse systems, on-farm reservoirs, etc.). Second, averages disguise high variability. Low-yielding and tail-end farmers typically pay a higher proportion of net income in water charges (Carruthers *et al.*, 1985). Figure 2.6 shows, for a sample of 101 rice farmers in Sri Lanka studied by Hussain (2005), that water charges would greatly decrease income for the 25–30% of poorer farmers even if, on average, they are only 10–15% of the average net income (Rs 11,000/acre).

In some countries, charges are limited by law in terms of either a maximum share of net income or another measure (e.g. Vietnam); in Iran, regulated surface water charges are limited to 1–3% of the gross value of crop output (Keshavarz *et al.*, 2005); in Cyprus, the charge is limited to no more than 40% of the weighted average unit cost (65% in exceptional cases) (Tsiourtis, 2002); in India, a 1972 policy review recommended that water rates should lie within the range of 5–12% of gross farm revenue (Prasad and Rao, 1991; Vaidyanathan, 1992). Elsewhere, minimum values are sometimes (ineffectively) decreed as in Korea (Sarker and Itoh, 2001) and Peru (Vos, 2002). Block tariffs have been proposed to protect the poor though others conclude that water pricing mechanisms are ineffective in redistributing income, besides having perverse subsidy effects (Tsur and Dinar, 1995; Dinar *et al.*, 1997).

PRICING AS A FINANCIAL INSTRUMENT: COST RECOVERY

Arguments for Cost Recovery

Funds for physical sustainability

The least controversial – and most compelling – argument in favour of cost recovery in irrigation is to ensure the availability of funds needed to sustain physical sustainability of

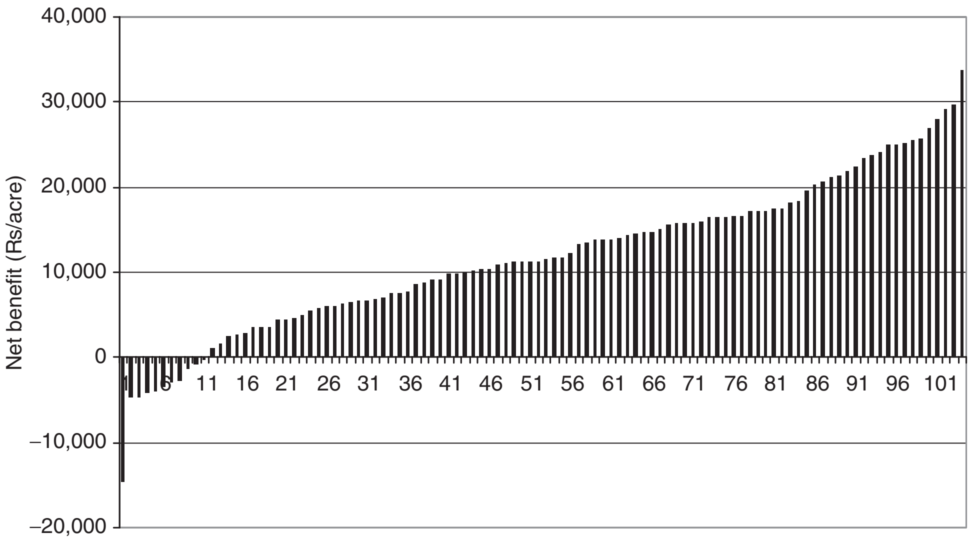

Fig. 2.6. Distribution of net income from rice cultivation (southern Sri Lanka).

the infrastructure. Concerns relating to physical sustainability have a long provenance, but rose to particular prominence in the 1980s when many governments and lending agencies faced the necessity of rehabilitating schemes that had sometimes been constructed only a few years back, but were already in a dilapidated state.[9] In Indonesia, for example, one-third of the 3 million ha of public sector irrigation schemes has been rehabilitated twice in the last 25 years (World Bank, 2005b). In the Philippines, successive projects funded by the World Bank and ADB have similarly returned repeatedly to the same national irrigation systems (World Bank, 1992) and, no doubt, other examples could be quoted. The decay of irrigation infrastructure leads to poor water delivery and is thought to lower agricultural production and decrease farmer income (Tiwari and Dinar, 2001; Hussain, 2005).

Degradation of facilities can be linked to many causes, including faulty design, shoddy construction, lack of incentives to respect covenants, pressures on public finances and a tendency by politicians to adopt a 'build-and-forget' approach to politically motivated projects. Widespread reliance on government for financing O&M has, in practice, led to underinvestment, deferred maintenance and degradation of facilities. This can also be related to 'public goods' and 'freerider' issues, as farmers intervene in low-level public infrastructure to secure their individual interests and as the incentives facing ill-paid operators and farmers have proved unsuited to the effective maintenance of both public and communal facilities. In many countries, tertiary maintenance is the responsibility of the farmers, yet even this is often poorly undertaken, in part due to the inability of the main system to guarantee predictable supplies, and in part due to lack of cooperation, freeriding and incentive issues at farmer level.

Underinvestment in maintenance is believed to be very considerable. For instance, total O&M requirements for public systems in India have been assessed at about Rs. 25–30 billion per year, yet less than a quarter of this amount is actually provided, with wide variation across states (Thakkar, 2000) and revenue receipts covering only 10% of expenditures in 2000 (Sur and Umali-Deininger, 2003). In Egypt, a desirable level of expenditures on O&M/rehabilitation has been put at US$234 million, yet only US$164 million is provided (Bazza and Ahmad, 2002). Comparable situations are found in numerous other countries, contributing to the perceived need for repeated rehabilitation as in Indonesia and the Philippines. The conclusion is that states have been de facto major defaulters and that sustainability depends on users taking over responsibility for maintenance.

Performance incentives

But paying for water does not by itself ensure good maintenance and service. When the receipt from water charges is channelled to state coffers, farmers come to regard charges as a tax rather than a direct benefit to themselves and pressurize politicians to reduce – even abolish – them. The assumption that paying for water in itself creates a sense of ownership has thus no doubt been overstated (e.g. Onjala, 2001, for Kenya).

[9]The literature provides uncontroversial evidence that these financial difficulties have been the driving force – or at least the chief justification – behind the revision of pricing policies, and also of many programmes of participatory irrigation management and varied degrees of turnover of management to farmer collectives (Frederiksen, 2005): see Burger, 1998 on Kazakhstan; Çakmak *et al.*, 2004 on Turkey; USAID, 2002 on Egypt; and Rap, 2004 on Mexico. Yet, the rhetorical argument that O&M costs are a 'huge drain' on state coffers appears frequently at fault. In 1997/98, canal irrigation subsidies were equivalent to 2.6% of the fiscal deficit in Karnataka and 7% of the fiscal deficit in Andhra Pradesh, with the same order of magnitude for Maharashtra, Rajasthan and Uttar Pradesh (Sur and Umali-Deininger, 2003). This seems significant, but only amounts to 0.1–0.3% of the respective state expenditures, a limited subsidy if redistribution to farming populations is considered a state policy (Molle, Chapter 5, this volume, and Venot *et al.*, Chapter 10, this volume, provide other examples for Thailand and Jordan).

When incentives are provided to the officials of the relevant organizations or to private subcontractors (these incentives may or may not be passed to users) to encourage collection or improve water management within the area they control, a link is established between payment and benefits to users. In order to close a virtuous circle of incentives, managers should ideally depend financially on farmers' contribution. Another fraction of the charges can be managed internally by a local group – e.g. farmers along a distributary or minor – for local repairs and maintenance or to pay ditch riders, thus ensuring that user payments are used to maintain the infrastructure and improve operations in direct sight of the farmers concerned. The focus here is not on paying benefit taxes to the state, but on ensuring both financial and physical sustainability through direct farmer involvement.

In sum, there are numerous variations of incentive mechanisms, depending on the degree of farmers' involvement in planning, allocation and hiring of staff, the level at which the boundaries are drawn between farmers' and agencies' responsibilities, and the inbuilt accountability mechanisms and incentives for financial contribution. Cost recovery makes full sense when arrangements are centred on financial autonomy, a clear definition of the responsibilities of managers and users and inbuilt accountability mechanisms (Small *et al.*, 1986; Small and Carruthers, 1991; Vaidyanathan, 1992; ICID, 2004; see Molle and Berkoff, Chapter 1, this volume, for a historical perspective). A reassessment of this model of financial autonomy will be attempted in a later section.

Equity considerations

Another important argument for recovering costs from farmers is that, having benefited from exceptional public investments, farmers should repay at least a part to the national budget on equity grounds (World Bank, 1984; Perry, 2001a,b). One mechanism for achieving this is a betterment levy (e.g. by increasing the land tax); another is by levying water charges. The equity argument is often supported by pointing to differences between investment in irrigated and rain-fed agriculture, and by the fact that water charges are seldom more than 5–15% of the incremental value of production relative to that of rain-fed output (Easter and Liu, 2005). Ministries of agriculture and irrigation typically spend much of their budget on irrigation (60% in the case of Thailand) and annual irrigation subsidies are often massive (Rosegrant, 1997; Sur and Umali-Deininger, 2003). Investment opportunities in rain-fed areas are no doubt more limited than in irrigated areas and it is perhaps understandable that governments start by developing regions that lend themselves to irrigation. Nevertheless, as argued earlier, irrigation subsidies have probably discriminated against the rain-fed farmer (ICID, 2004).

A related equity argument is that cost recovery can contribute funds for irrigation expansion in currently deprived regions, an argument notably employed by politicians in advocating investments in their constituencies[10] (World Bank, 1984) and by those who advocate irrigation as the driving force for regional development. However, if income from water charges or betterment levies is accrued to the general public budget, there is no assurance that it will be used to expand irrigation since Ministries of Finance typically allocate resources in line with general political priorities.

Objections to Cost Recovery

Identification of beneficiaries

At first sight, it is obvious that farmers are the beneficiaries of irrigation and the large majority welcome irrigation projects. Even so, they are neither consulted on construction nor are their obligations always clearly defined. Some may have to relinquish land while others may have invested earlier in private or communal

[10]This may unfortunately lead very often to uneconomic projects which are granted against political support to the ruling party, or to other MPs ('pork barrel' in the USA). A perverse outcome can be the 'overbuilding' of river basins (Molle, 2007).

irrigation and gain little by being included in the new scheme (e.g. in Iran, Thailand or Argentina). Demanding repayment of costs decided by the state in these cases seems inequitable. Moreover, irrigation is often provided in the context of multi-purpose projects and irrigation itself may benefit non-farmers (e.g. domestic users or those in the flood plain). Since cost allocation is seldom applied systematically, irrigators may be asked to pay more than a fair share of joint costs (though hydropower rather than irrigation is more typically overcharged). Moreover, as argued earlier, if much irrigation is underpinned by strategic objectives and is inherently uneconomic, recovery of full costs is neither fair nor practicable: 'Is it fair to charge the full cost (including the capital cost) for projects designed without the farmers' say or designed on the basis of higher world grain prices?' (ICID, 2004).

Cost recovery is sometimes taken to imply that all costs should be recouped from direct beneficiaries. However, some argue that the 'joint private/public nature of benefits that result from such projects' and the long-term nature of economic returns may warrant subsidization by the state (Kulshreshtha, 2002). Others assert that irrigation facilities are a form of social overhead capital with farmers being just one category of beneficiaries amongst many (Small, 1996). If so, it is arguable that other beneficiaries – traders, processors and transporters – should be charged a share or irrigation costs. More broadly, a whole region may benefit from the stimulus of irrigation and consumers everywhere benefit from rising farm output in the form of lower prices (Sampath, 1992; Small, 1996; Bhattarai *et al.*, 2003). Thus, it is sometimes argued that 'indirect beneficiaries of irrigation, (notably) consumers of cheap food, should be happy to subsidize irrigation development through taxes' (Perry, 2001a,b).

Care must be taken in disentangling these arguments. If multiplier benefits are limited to *incremental* impacts relative to those of the alternative project (which also, invariably, exhibit such multiplier effects), then – for this and other reasons – the conditions under which they can be included in total benefits are restrictive (see first section). Moreover, food marketing is often amongst the most competitive sectors in developing countries. If so, participants, by definition, pay almost full economic costs so that charging specific indirect beneficiaries for a share in irrigation costs risks double-counting. The justification given for indirect benefits is thus less convincing than sometimes implied.

As Abu-Zeid (2001) recognizes, governments may 'continue to subsidize [new] projects for several reasons, e.g. enhancing national security, maintaining political stability, decreasing population density in certain sensitive geographical regions and conserving water'. Given these national objectives, the level of capital cost recovery that is desirable is ultimately a political judgement given the context concerned, reflecting judgements on the weights given by society to national objectives other than economic optimization.

Cost estimation

Cost estimation – and hence the level of cost recovery implied – is seldom straightforward. For schemes constructed in part with unpaid labour (whether voluntary or otherwise) – as in China, Vietnam, Burma and at the tertiary level in many countries – implicit farmer contributions should be excluded. FAO and USAID (1986) have also suggested that 'farmers should not be asked to repay the cost of over-elaborate gold-plated designs, incompetent, expensive construction, costs overruns for reasons of corruption, bad scheduling of construction activities or the like'. Similarly, farmers should not be asked to pay for overstaffing,[11] poor management and corruption (Rao, 1984; FAO and USAID, 1986; Bhatia, 1991; Gulati and Narayanan, 2002 – Rao has estimated that in India only about half of officially estimated costs represent real costs). Moreover, with regard to maintenance, should actual costs or ideal costs be

[11]Lee's (2000) review of 82 irrigation providers found an average of 38% of O&M costs spent on salaries, with a maximum of 82%; it is 80% in Sindh, Pakistan (SIDA, 2003), but only 10% in northern Vietnam (see Fontenelle *et al.*, Chapter 7, this volume).

considered and how should the ideal be defined? Systematic maintenance may lengthen a project's life, but what is the economic optimum? Finally, convincing farmers that opportunity and externality costs are real, let alone charging them for these costs, is extraordinarily difficult (see later section).

Irrespective of whether actual O&M and related costs are justified, they must be financed either by government or by farmers if irrigation is to be sustained. As noted earlier, scheme autonomy strengthens incentives for containing costs to those justified by prevailing conditions. In the state of Victoria, Australia, for example, when farmers were required to pay the full costs of O&M, increased scrutiny of the supply agency led to a 40% reduction (World Bank, 2003a,b). While farmers tend to take a short-term view of what is required, often in the hope that government will, in due course, rehabilitate the scheme, they also usually have a much better idea than unaccountable public agencies of what is truly required (sometimes less than external experts commonly suppose).

Cost Recovery: Empirical Evidence

The literature suggests that no more than a portion of O&M costs is typically recovered (Dinar and Subramanian, 1997; Cornish *et al.*, 2004; Easter and Liu, 2005), a conclusion that probably holds despite inconsistencies in the definition of these costs. OECD countries often recover full O&M costs (Garrido, 2002; Berbel *et al.* Chapter 13, this volume), while Latin America (notably after management transfer) and the Mediterranean basin (e.g. southern Europe, Tunisia, and Morocco) have fared better than Asia and Africa, and East Asia better than South Asia (ESCWA, 1999 for Western Asia; Ringler *et al.*, 2000 for Latin America; Chohin-Kuper *et al.*, 2002 and Bazza and Ahmad, 2002 for Mediterranean countries; Cornish *et al.*, 2004 for a review). Figure 2.7

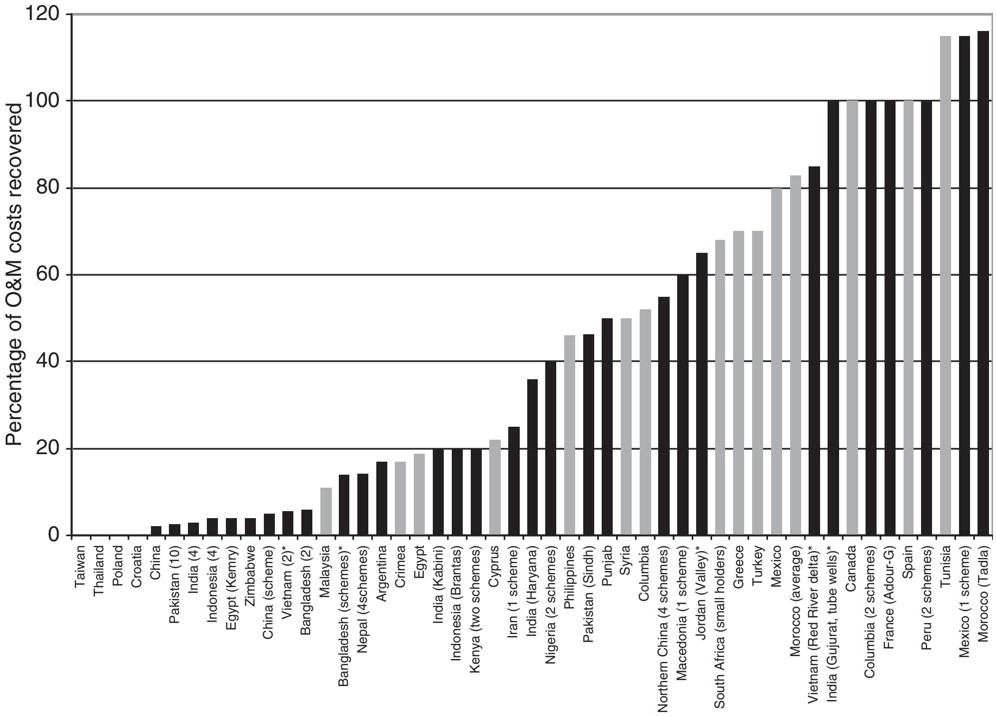

Fig. 2.7. Water charges relative to O&M costs in selected schemes and countries.

plots average levels of cost recovery for a number of cases, distinguishing between particular schemes (both gravity and pressurized marked with*) and country averages (in grey).

Beyond these average estimates drawn from the literature, in practice both O&M costs and cost recovery levels vary over time depending on water use patterns and the age of systems, government policies and organizational arrangements (Carruthers *et al.*, 1985). For instance, the real irrigation charge in Tunisia was raised by 2.4 times between 1990 and 2000 and collections rose from 57% to 90% so that they now cover, on average, 115% of O&M costs (Hamdane, 2002a,b). In Morocco, charges in the Tadla scheme cover both O&M and depreciation (Hellegers *et al.*, Chapter 11, this volume), although they cover no more than O&M costs in three other gravity schemes, and 66% in three major pumping schemes (values for 2001; Belghiti, 2005b).

Historical evidence suggests that in no country have the beneficiaries shouldered a significant share of the initial capital costs of large-scale irrigation, let alone the costs of subsequent irrigation expansion. Many schemes date back to when irrigation expansion was a national policy and are targeted for cost recovery mainly to contain current public expenditures. Even in richer countries, it is difficult to justify the recovery of capital costs of past public projects, given that irrigation benefits have usually been capitalized in land values and, given that relative price shifts often make it financially impossible (see Pigram, 1999 on Australia; Musgrave, 1997). Postel (1992), for instance, reports that 4 million ha in the west USA are supplied 'at greatly subsidized prices' by the Federal Bureau of Reclamation (see also Anderson and Snyder, 1997), reflecting the fact that the 1902 legislation emphasized western settlement rather than full market returns for Federal water projects (Gollehon *et al.*, 2003). Irrigators in the Central Valley Project have repaid only 4% of the capital cost. Currently, repayment of capital costs averages about 15% in real terms (Howe, 2003; Hanemann, 2006).

In South Korea, financially autonomous Farmland Improvement Associations (FLIAs) have repaid part of initial capital costs, in addition to shouldering full O&M costs (ADB, 1986b) and in Japan corporate Land Improvement Districts shoulder 10–15% of the costs of large-scale state irrigation projects and 25% of medium-scale projects initiated by prefecture governments (Sarker and Itoh, 2001).[12] The principle of capital cost recovery has been incorporated in European directives and has the clear potential to ensure that projects are cost-effective and to crowd off marginal and politically motivated water resource development (Garrido, 2002). Yet, perhaps for this very reason, obstacles still prove pervasive and fiscal discipline elusive (Hill *et al.*, 2003).

Morocco is a rare example in the developing world in having an Agricultural Investment Code that specifies 'with the objective to alleviate the [financial] burden on farmers, (irrigation rates) will be called upon to contribute to investment costs *only* to the level of 40% of these costs' (Belghiti, 2005a; emphasis added). Although this level has yet to be attained Morocco has taken bold steps towards financial autonomy. In Egypt, new irrigation areas (New Lands) for commercial entrepreneurs are also being granted with a degree of cost sharing (Perry, 1996), while expansion of the irrigated area in the Office du Niger (Mali) included 20% of contribution by farmers (Aw and Diemer, 2005). In contrast, in Bihar and Haryana, where irrigation remains firmly in the public sector, if capital costs were charged in full, payments would amount to 40–90% of net incremental farm income (Bhatia, 1991).

Development agencies have long been reluctant to recognize that few countries will recover more than a nominal share of initial costs, and that irrigators' 'debt' to the state will be eventually written off, even in developed countries (Garrido, 2002). For example, ADB's 1985 review (ADB, 1986a) calls for 'benefit-conscious project preparation' and notes that the disregard for loan covenants

[12]It is perhaps no coincidence that South Korea and Japan simultaneously subsidize their rice-farming sector through import duties and controls that lead to very high internal prices and promote domestic production.

(in particular on ISFs) by governments is not being addressed. Pitman (2002) observes that 'Globally, most [World] Bank projects pay lip-service to (capital cost) cost recovery', but that those which addressed this issue in practice were largely water supply projects. Recognition of the case against full capital cost in irrigation and greater realism in practice would clearly be desirable (World Bank, 2003a,b).

Empirical evidence also shows that very seldom are incentives linked to charges. Bos and Wolters' (1990) survey of 159 schemes covering 8 million ha showed that there is no relation whatsoever between the level of charge and efficiency. This was confirmed by later findings by Jones (1995) which showed that revenue from water charges generally goes to the general treasury and is not earmarked for O&M. A typical example is Pakistan where revenues from water charges go to the provincial or state treasury, losing the link between payment and O&M and quality of service (Bazza and Ahmad, 2002) (see also Jordan: Venot *et al.*, Chapter 10, this volume; and India: Samal and Kolanu, 2004). Conversely, the failure to ensure reliable supply is one of the major reasons for widespread defaulting (Carruthers *et al.*, 1985; ADB, 1995; Spencer and Subramanian, 1997). Samal and Kolanu (2004) note the 'categorical and explicit refusal of [Indian] farmers to pay the water tax till the irrigation service was improved'. In Sindh, Pakistan, 'farmers are not willing to pay since the financial system is not transparent and they do not see that the charges paid are used to deliver a good service'. The farmers said that they were willing to pay for services, but not for 'someone's wife's jewellery' (Cornish and Perry, 2003).

Even where progress has been made in transferring responsibilities at the tertiary or secondary level to farmer organizations under irrigation transfer and similar programmes, supply has often remained unpredictable. Whether due to suboptimal management, to real constraints in controlling stochastic water variability and uncertainty or to what happens upstream, insecure main system supplies have undermined efforts by farmers to organize at secondary or block level. For example, Parthasarathy (1999) has shown that, in Gujarat, India, WUA members failed to pay higher rates when they appreciated that managing an isolated or terminal portion of the canal system failed to contribute to any real improvement in the reliability of water supplies. As Freeman and Lowdermilk (1991) put it: 'To disconnect farmer payments of assessment for maintenance, whether in cash or kind, from water delivery is virtually to invite organizational decay.'[13]

In most countries, governments continue to be responsible for the funding of main-system O&M, together with replacement, rehabilitation and modernization works, quite independently of charge collection itself. In other countries, notably in East Asia, Latin America and much of North Africa (as well as in most developed countries), irrigation water charges are collected and retained by scheme management (irrigation district). But even in these situations, O&M expenditures can be deficient. In China or Vietnam, for instance, the level of water charges is regulated by national, provincial and local price commissions, and, though in principle authorized charges are based on estimated requirements, in practice increases have been limited with a view to reducing burdens on farmers (Hydrosult, 1999; Lohmar *et al.*, Chapter 12, this volume). Similarly, the Government of the Philippines has repeatedly failed to authorize the NIA to effectuate needed increases in water charges (World Bank, 1992). Financial autonomy – total or partial – has been practised widely in developed countries,

[13]In addition to farmers' reluctance to contribute, low rates of recovery are compounded by agencies' reluctance to enforce collection (Carruthers *et al.*, 1985), due to drudgery avoidance, unwillingness to antagonize farmers and desire to keep good relations, sympathy for their economic situation, or fear to give farmers reasons to question the quality of service.

including the USA, Spain, France, Italy, Mexico, Japan and Korea.[14]

PRICING AS AN ECONOMIC INSTRUMENT: WATER CONSERVATION

Introduction

That water is wasted due to underpricing is a widely held view, from the former President of the World Bank ('the biggest problem with water is the waste of water through lack of charging': Wolfensohn, 2000) to the World Water Vision ('users do not value water provided free or almost free and so waste it': Cosgrove and Rijsberman, 2000), to detached analysts ('water is consistently undervalued, and as a result is chronically overused': Postel, 1992) and environmentalists who favour 'developing a pricing system that prevents excessive use of water' (WWF, 2002). For the EU (2000b): '[E]fficient water pricing policies have a demonstrable impact on the water demand of different uses. As a result of changes in water demand, efficient water pricing reduces the pressure on water resources. This is particularly true for the agricultural sector.'[15]

Seemingly corroborating the assumption of waste is the fact that irrigation accounts for approximately 70% of withdrawals on average. Agriculture 'gobbles up at least 75% and sometimes as much as 90% of the available water', while 60% of water deliveries fail to reach the fields (The Economist, 2003). Profligacy combined with agriculture's dominant share suggests an easy solution: if raising irrigation charges can reduce losses even by a small percentage, sufficient water can be freed to meet the much smaller demands of other expanding sectors (World Bank, 1993; Winpenny, 1997; Gleick, 2001; Louw and Kassier, 2002; Davis and Hirji, 2003; IRN, 2003).

This section evaluates whether low water charges lead to waste and higher charges promote conservation. It first examines the received wisdom that 'water is wasted because it is underpriced'. Then it examines the conditions under which pricing water can be a 'key to saving water' and assesses the empirical evidence. It concludes by evaluating the potential of pricing for promoting conservation.

Is Water Wasted Because It Is Underpriced?

Is water wasted?

The first section showed that the concept of irrigation efficiency is often misstated. If water is abundant – in surplus basins, or during the rainy season, after it rains – excess diversions matter little since they return to the hydrological cycle (though, of course, they can impact adversely on water control, waterlogging and flooding). If water is scarce, farmers compete for the limited flows available: the struggle for water *when it is scarce* means that little water is wasted *when it has value*, and this is shown by observation of shortage situations. Moreover, losses may be used – after a delay – downstream or from aquifer recharge and only if water flows to the sea or another terminal sink is it no longer available for human use.[16] The central issue is thus one of *basin* efficiency and focusing on farm-level or scheme efficiency can be very misleading.

[14]Although this autonomy is partly paralleled with, or allowed by, massive subsidies granted through output prices or direct payments.

[15]See also 'Inefficient pricing and management of irrigation water supply leads to massive wastage' (Hansen and Bhatia, 2004) and similar statements in Holden and Thobani (1996), FAO (1998), ESCWA (1997), UNESCAP (1996), Ringler *et al*. (2002), TDRI (1990), Siamwalla and Roche (2001), Roth (2001), Bate (2002), etc.

[16]Flows to the sea may still, of course, have important environmental functions, including: flushing out sediments, diluting polluted water, controlling salinity intrusion and assuring the sustainability of estuary and coastal ecosystems.

There might be cases of a water-abundant scheme located within a water-short basin. Such a situation may be due to locational reasons, specific water rights or political influence that insulates that particular scheme from overall scarcity. This is a problem of (basin-wide) allocation and equity, which has other roots and will not be solved by pricing policies.

Is wastage due to low prices?

The above explanation implies that much less water is 'wasted' than is commonly supposed. Residual 'real' losses (evaporation from open surfaces, transpiration via unproductive growth, etc.) may be identified on a case-by-case basis but can 'real' losses be attributed to low water prices? A first issue is that shifts in farmer behaviour (induced by prices or otherwise) only impact on the share of diversions they receive. Ray (Chapter 4, this volume), for instance, estimates that farmers in the Mula scheme receive no more than 30–35% of the water released from the reservoir, the remainder being 'lost' from the canal system. Typical losses of 50% imply that raising the water charge to farmers can at best impact on about one-half of the water diverted. A second issue is that scheme-level deficiencies primarily relate to inequities (head-end and tail-end problems) and socio-economic costs rather than physical losses. Whenever wastage (or shortage) occurs, it is because the supply made available at the farm inlet is not in line with needs, and the causes of this mismatch remain largely independent of the users themselves (Grimble, 1999; Rodgers and Hellegers, 2005). Resolving such problems is primarily an issue in design and management, and remedies lie at the system level rather than with changing the behaviour of farmers (Chambers, 1988): effective control of supply is needed but, as Small (1987) aptly observed: '[I]t is likely that once this prerequisite exists, the amount of "wastage" will be greatly reduced, thus lowering the potential efficiency gains from any subsequent attempt to introduce water pricing.'

Conditions for Water Pricing to Elicit Water Savings

Although the causal relationships between low water-use efficiency and low prices are weak, and the fundamental objective is to optimize agricultural returns rather than minimize physical losses for their own sake, there is nevertheless a case for adopting pricing policies whenever they can contribute to this fundamental objective. Although the opportunities may be very limited, there is a continuum from conditions where price has no impact on water use and solutions lie entirely in management, to conditions where water is on demand and farmers can adjust volumes to reflect marginal returns (Fig. 2.3). This subsection addresses the prerequisites for the latter (see also Ray, Chapter 4, this volume). Associated issues related to externality and third-party impacts are considered in a later section.

Is pricing volumetric?

It is sometimes argued that, by making farmers aware of the value of water, even a flat rate promotes water savings (for Tanzania, see van Koppen *et al.*, Chapter 6, this volume). But there is little evidence for this: on the contrary, farmers try 'to get as much as possible of the thing for which they have been taxed' (Moore, 1989; Bos and Wolters, 1990; Berbel and Gomez-Limón, 2000).

Pricing can thus conserve water only if supply is volumetric. Problems of volumetric measurement are well known (Moore, 1989; Sampath, 1992; Rosegrant and Cline, 2002). For historical, technical, financial and managerial reasons, measurement at farm level is rare and even then charges may not be based on measured volumes. In some cases (e.g. for paddy), measurement at the farm level is unworkable without major structural investment (Moore, 1989) and installing functional devices in flat gravity systems (e.g. in deltas) is impracticable. More generally, measurement at the farm level is prohibitively expensive in surface systems with thousands, if not hundreds of thousands, of small farms. Tampering is pervasive and the transaction costs of data collection, monitoring and enforcement are beyond the capacity of most agencies and control at farm level is an illusion: Cornish *et al.* (2004) conclude that 'in practice, volumetric methods of supply to individual farmers are probably not feasible in large parts of the developing world at present'.

Charging for bulk allocations – to a WUA, distributary organization or other scheme entity – is a way to circumvent the transaction costs of charging for individual supply (Carruthers *et al.*, 1985; Repetto, 1986; World Bank, 1986; Asad *et al.*, 1999) and is needed in any case for effective (volumetric) management. But, if bulk charges are to impact on water use, contractual or quasi-contractual agreements must be enforced (Fig. 2.3) which requires more than reforms based on little more than wishful thinking, as noted earlier. While enforcement and collection delegated down the system, closer to the farmer tends to promote participation and accountability, the critical point is to pass incentives on to farmers.

Is water demand elastic?

A second obstacle to effective conservation pricing is that the elasticity of demand for irrigation water at current charges is low or negligible (de Fraiture and Perry, Chapter 3, this volume). Bos and Wolters (1990) found that in all but one of the projects studied charges were less than 10% of net farm income and 'too low to have significant impact'. Latinopoulos (2005) found no relationship between charges and water use in a sample of 21 irrigation districts in Greece, and a study of nine Spanish schemes attributed differences in water use to other factors (soils, nature and abundance of the source, history, etc.), concluding that inelastic demand reflected the relatively low share of water in production costs and the lack of a substitute (Carles *et al.*, 1999). Some studies carried out in the USA indicate a similar lack of responsiveness to price (Hoyt, 1982; Moore *et al.*, 1994). Volumetric pricing is most often associated with pressurized systems and high-value crops, the very situations where efficiency is already high and water costs (hence elasticity) marginal (Albiac *et al.*, 2006).

That volumetric charges seldom impact significantly on farmer behaviour (Gibbons, 1986; Malla and Gopalakrishnan, 1995; Bosworth *et al.*, 2002; Rosegrant and Cai, 2002) is perhaps hardly surprising given that irrigation water is a subsidized intermediate input. There is probably always a range over which demand is elastic, with elasticity rising as charges approach full cost. However, such charge levels have been shown earlier to be unrealistic in uneconomic schemes where water is subsidized. At current levels, even large increases make little impact since other costs are relatively more important, and cross-elasticities determine water use. Water prices in Iran, for instance, would need to rise by a factor of 10 to be effective in curtailing demand (Perry, 2001). Given the political sensitivity of pricing issues governments cannot be expected to risk raising charges well above O&M costs, just for the sake of encountering elasticity.[17]

In contrast to inelastic demand at farmer level, autonomous irrigation entities should, in theory, behave like profit-maximizing industries and reduce use in response to all bulk charges. In developed countries, regulators require irrigation districts to cover costs but even then they often skimp on O&M and/or seek other income sources to avoid 'bankruptcy'. In developing countries, farmer resistance to enhanced charges is stronger, whether the system is managed by government agencies, canal organizations or WUAs. Evidence from China and elsewhere

[17] Although this is advocated by Brooks (1997): 'Most would argue that . . . water tariffs should be designed to encourage conservation, not just to recover costs (which implies that pricing should be high enough to move into the elastic portion of the demand curve).'

(see below) suggests that institutional reforms can strengthen main-system management and transfer costs to autonomous entities, but there are still few examples where bulk water charges as such have led to significant water savings.

Lastly, true elasticity of response is very hard to establish because there is so little information on the relationship between improving efficiency at the farm level and the costs of doing so for a given irrigation technology and a given pattern of supply (see de Fraiture and Perry, Chapter 3, this volume). All shifts involve costs, e.g. in increased drudgery, labour or capital, and depend, *inter alia*, on farmer strategies and on the opportunity cost of their labour[18] (Venot *et al*., Chapter 10, this volume); but estimating such costs and the associated responses is complex. Modelling exercises almost invariably oversimplify and focus on induced changes in terms of crop mix or technology without recognizing all the costs involved. As a result, the estimates of elasticities tend to be crude and unconvincing (more on this later).

Water Pricing and Water Savings: Empirical Evidence

Dinar and Subramanian's (1997) cross-country review showed that water prices across countries are not related to relative water availability, suggesting either that the current objective for charging is not to manage scarcity, or that other factors come into play. That countries with higher scarcity are not 'more aggressive in reforming pricing schemes' also brings out that other mechanisms are preferred. This was confirmed by a 2000 review of the last 67 irrigation projects funded by the World Bank, which revealed that in none of the projects had water charging mechanisms been planned as incentive tools (Tiwari and Dinar, 2001). Since, in any case, relations between water use and prices can only be expected under conditions of volumetric management, we focus here on cases of bulk allocation and individual volumetric pricing.

Bulk allocation

Sri Lanka, Turkey, China and Mexico are amongst countries that have promoted bulk allocation and in some cases have also introduced charges for bulk supplies:

- Evidence from Mahaweli System H in Sri Lanka showed that allocation at block level can lead to lower diversions, but this is primarily due to stricter scheduling and improved main-system management, resulting in more predictable and uniform flows and reduced conflicts. Water charges are not differentiated at farm level, and though WUAs are charged in proportion to water allocations, charges are not based on volumetric measurement and are too low to provide incentives for water savings (IWMI, 2004).
- Similarly, in Turkey, major irrigation has largely been transferred to irrigation districts that receive bulk water at no cost though they are expected to meet O&M costs in their own area. Reliability of supply has improved and fee recovery has increased substantially (Yercan, 2003; Özlü, 2004), the transfer of the financial burden of O&M to farmers being the main objective of the programme (Ünver and Gupta, 2003). But flat-rate charges have no impact on water conservation at farm level and tertiary distribution remains deficient (Yercan, 2003).
- The transfer programme in Mexico goes a step further (Kloezen, 2002). The

[18]Such interventions include avoiding breaches in bunds or continuous irrigation (for rice farmers), fine-tuning cut-off time to avoid losses at the end of furrows or not using sprinklers on windy days. Other adjustments relate to changing cropping techniques, like resorting to rice dry-seeding (e.g. in the Muda scheme, Malaysia: Guerra *et al*., 1998), using mulch in vegetable plots or reducing the length of furrows. Other responses are more capital-intensive, such as laser land-levelling, which allow reduced and more homogeneous application of water by gravity, and frequent renewal of drippers in micro-irrigation.

National Water Commission in consultation with user representatives determines allocations to Irrigation Districts on an annual or seasonal basis. Bulk charges are met out of an O&M charge assessed and collected by WUAs and passed to the Commission via the District. Although O&M charges are levied in proportion to the amount contacted to the farmer by the WUA, they remain fairly low (2–7% of gross product in the scheme studied by Kloezen) and reflect O&M costs rather than conservation objectives. Seasonal quotas are tradable amongst WUAs within a district, with trades usually triggered when a WUA cannot meet the contractual demands of their members (Kloezen and Garcés-Restrepo, 1998). Maintenance is often suboptimal, with many WUAs unwilling to incur major costs and raising revenues only as immediate needs arise (Pérez Prado, 2003).

- Lessons from China are masked by the diversity of physical and institutional settings (Lohmar *et al.*, Chapter 12, this volume). Water is usually delivered in bulk by basin and system organizations to township or village entities, WUAs and even private operators. Bulk water charges in some cases have contributed to reduced diversions as entities at each level seek cost savings. Generally, however, even if bulk water supplies are priced volumetrically, current pricing policies rarely effectively encourage water saving at farm level (see Fontenelle *et al.*, Chapter 7, this volume), in part because farmers may be unaware of how water charges relate to other rural charges. Farm quotas necessarily decline when diversions decline but the reform process still appears strongly government-controlled (Mollinga *et al.*, 2005).

These examples confirm that bulk allocation is primarily a mechanism for: (i) improving the predictability and reliability of deliveries at basin and main canal levels; and (ii) allowing partial financial and managerial autonomy to WUAs, thus shifting part of the O&M costs to them. Bulk water pricing can generate revenue, but even if farmer charges are assessed in relation to delivered quantities, they are seldom charged on a volumetric basis; and even if charged volumetrically, they are seldom high enough to promote conservation (Asad *et al.*, 1999; Tiwari and Dinar, 2001). Internal trading (as in Mexico) can improve scheme-level efficiency but, of the examples quoted, only in China is there evidence that some scheme managers have a clear incentive to reduce bulk diversions (Lohmar *et al.*, Chapter 12, this volume).

Individual Quotas and Irrigation on Demand

Technical control may allow volumetric monitoring at farm level, but only if water is supplied on demand can the full potential of water pricing be realized. There is a continuum from individual quotas to irrigation fully on-demand, depending on how constraining quotas are and how responsive the system is to user requests:

- In Morocco, farmers pay a minimum fee equivalent to 3000 m^3/ha (Ait Kadi, 2002). In most cases, water is distributed by rotation and farmers must pay the full amount. In practice, quotas are low and any savings would depend in effect on the adoption of micro-irrigation. The water charge is based primarily on cost recovery rather than on conservation criteria, though in pump schemes the water bill can be up to 65–70% of gross income (e.g. Souss Massa groundwater: Ait Kadi, 2002) and in these cases it undoubtedly influences farmer behaviour.
- In Jordan, quotas in the valley are assessed at individual level and based on crop type, thus promoting water savings (Venot *et al.*, Chapter 10, this volume). Despite pressurized systems over most of the area, water variability and canal capacity preclude arranged demand irrigation and water is rotated at block level. Charges are set in relation

to O&M costs rather than to regulate use, though higher charges may prompt crop shifts and raise water productivity. The (coming) Wahda dam (Courcier *et al.*, 2005) and on-farm reservoirs help offset the rigidities of rotational delivery.

- European countries – Italy, France, Spain – also provide examples of modern pressurized irrigation systems that handle scarcity in the first instance by quotas (which may be very low, e.g. 2000 m^3/ha in Capitanata (South Italy), Genil Cabral (Spain) and the Neste system (France)).[19] There is usually flexibility at the margin with the above quota-use penalized at rates as high as 10 times the variable component in Charentes in France, and 25 times unit cost in Genil Cabral (Maestu, 2001; Montginoul and Rieu, 2001). Water distribution is usually by 'arranged demand' rather than under direct farmer control, and rotational delivery is often required at peak periods or during droughts.
- In Israel, the small unified distribution system is almost fully reticulated and pressurized, and backed by storage in the Sea of Galilee and managed aquifers. In contrast to systems of 'arranged demand', cooperatives and farmers retain discretion over when to irrigate under normal conditions. However, they are subject to cooperative and/or individual quotas that are charged at rising block rates. This has contributed to regulating water demand at the margin (Kislev, 2001) so that average use has sometimes been below the quota. Quotas in principle are adjusted annually but, in practice, they are regarded as water rights (Plaut, 2000; Kislev, 2001).
- A system that comes close to fully on-demand is that operated by the Canal de Provence in France, where the main canal is dynamically regulated to meet agricultural and municipal demands. No formal quotas are announced and farmers are free to irrigate as they wish (although they have to subscribe to a given delivery discharge). Prices are set to recover costs rather than to control demand, but the price structure is complex (Jean, 1999), distinguishing differing periods and between peak and normal demand, and it can be assumed that there are some incentives for water savings.
- Other cases include California, Canada, Peru and China. During the 1990–1994 drought in California, Broadview's water supply had to be decreased by more than 50%. Instead of raising prices in order to reduce demand accordingly, it was found preferable 'to begin allocating water among individual farmers' proportionally to the size of their farms, while providing cheap loans to encourage farmers to purchase sprinklers and gated pipe irrigation systems (Wichelns, 2003). In one system of northern Peru studied by Vos (2002), pricing was volumetric but was not used to manage scarcity: rather in times of shortages the rules employed promoted equity and defined quotas that limited use. In Shangdong, China, the use of integrated circuit (IC) machines ensures that farmers cannot obtain irrigation water without paying (Easter and Liu, 2005) and seems to provide reliable on-demand water.
- In some countries (e.g. in western states of the USA, Chile, etc.) quotas are defined as individual rights and a legal framework has been developed for trading these rights. Management continues to be determined by quotas and water distribution is still, usually, by 'arranged demand'. However, water trading redistributes quotas and contributes to higher economic returns. System constraints, third-party concerns and regulatory aspects may confine trades to neighbouring farmers, with little impact on irrigation water use, but in some places water is traded out of agriculture (e.g. the Colorado-Big-Thompson scheme).

[19]See Mastrorilli *et al.* (1997), Altieri (2001), Berbel *et al.* (2001), Hurand (2001) and Maestu (2001).

Public and communal groundwater suffers many of the same constraints as surface irrigation. A study of collective wells in Mexico – which modelled crop and irrigation options – showed, for instance, that a 30% reduction in groundwater use would require water charges to be (unrealistically) raised by a factor of 4 (Jourdain, 2004). In contrast, private groundwater approximates to irrigation on demand. So long as groundwater is abundant and input and output markets remain undistorted, extractions are determined by costs or prices and the results can approximate to an economic out-turn. But, in contrast to surface systems subject to supply constraints and quotas, in the absence of these preconditions groundwater regulation is seldom feasible since the transaction costs usually prove insurmountable, given the number and dispersal of numerous small wells. Even where regulation is, in principle, feasible, for legal and historical reasons much groundwater continues to be unregulated.

Quotas versus Prices

Three main conclusions can be drawn from the above review. First, and most obviously, incentive pricing requires volumetric management and is thus precluded in the vast majority of developing country situations, at least at farm level. Second, even if volumetric supply is assured at farm level, in practice, price incentives are predominantly used at the margin to control use in excess of defined quotas or rights. This gives users some flexibility, whether water is distributed by 'arranged demand' or is under the control of users. This provides incentives for water saving, but falls short of true irrigation on demand. Third, even for systems that approach on-demand irrigation and have the capacity to meet peak demands, rights are capped by a quota and suspended (e.g. in favour of rotational distribution) during droughts since irrigation invariably receives low priority.

In other words, even in the rare cases where conditions are met to regulate demand through pricing, supply is instead invariably managed through administered quotas or water rights. Reasons for the predominance of quotas include: (i) transparency; (ii) ability to ensure equity when supply is inadequate; (iii) administrative simplicity and relatively low transaction costs; (iv) capacity for bringing water use directly in line with continuously varying available resources; and (v) limited income losses incurred (as compared with price regulation). 'When water is scarce, the surest and most common way to make customers use less water is to limit supply' (Cornish *et al.*, 2004) and this has been easily the most favoured solution for restraining demand (Bate, 2002).[20]

But quotas also have their drawbacks (Bate, 2002; Chohin-Kuper *et al.*, 2002; Tsur, 2005). While price or market regulation tends to promote economic efficiency at the cost of equity (Okun, 1975), quotas (when non-transferable) foster equity at the cost of efficiency: they can lack flexibility in response to changing circumstances, as in the case of settlement quotas in Israel.[21] Equity is also weakened in the case of conjunctive use of

[20] The virtues of rationing (in the short term) and/or the allocation of quotas (for long-term allocation) are getting more attention from the World Bank (2006) who reckoned that 'quotas work better than prices when water users are not very responsive to water price changes'. Bosworth *et al.* (2004) also concluded that 'getting the prices right' is not the most appropriate solution to managing scarcity.

[21] The Israeli case is instructive of the difficulty to readjust quotas once they have been defined and, at the same time, of the growing mismatch which can materialize between one village quota and its real use or needs (Plaut, 2000). The trajectories of kibbutzim and cooperatives depend not only on many factors, including ethnic composition, level of education and political linkages, but also on the links to markets, the availability of non-agricultural opportunities and the possible development of additional local resources (Lees, 1998). With time, some settlements (and some farmers within each settlement) tend to intensify agriculture, while others shift to partial farming. Resulting imbalances between quotas and needs have led to some inefficiency; in the 1980s, some farmers would irrigate carelessly so as to fully use their quota for fear of seeing it reduced (Lees, 1998); and trading within as well as between communities has emerged (Kislev, 2005).

canal water and groundwater, where quotas are rarely adjusted to rebalance overall combined supply (like in Morocco). In practice, quotas also often integrate pre-existing local systems of rights (see the Jordan valley in Venot *et al.*, Chapter 10, this volume). In the absence of an 'omniscient allocator', reallocation can be done either through rules that embody desired priority principles or by making quotas tradable, or by a combination of both in order to address equity concerns while promoting efficient allocations (Seagraves and Easter, 1983; Bjornlund and McKay, 1999; Johansson *et al.*, 2002).

It is true that management of quotas cannot fully simulate the economic scarcity signals of a market price. But, given the socio-economic and practical constraints, and the political costs of promoting irrigation pricing for managing scarcity, the management of quotas (the 'visible hand of scarcity') appears a far more satisfactory and practical solution to water savings in almost all real-life circumstances. Even in Europe, where pricing is being strongly promoted, Garrido's (2002) review concluded that 'irrigation pricing reforms should not expect significant reductions in farmers' water consumption' and that 'efficient allocation can be made without prices'. It should be noted that this conclusion does not rule out on-demand irrigation when feasible and cost-effective. Also, it does not rule out the development of regulated markets in water rights (or quotas) where willing buyers and willing sellers cooperate to transfer water from low-value to high-value uses (see later section).

PRICING AS AN ECONOMIC INSTRUMENT: CROP AND TECHNOLOGICAL CHANGE

Shifts in Cropping Patterns

Governments often seek to promote agricultural diversification. This may be to save water but the primary objective is to generally promote agricultural growth and raise farm incomes. Some equate the two, arguing that, if the price of water is raised (ideally to its opportunity cost), low-value crops are less attractive and farmers shift to higher-value crops (Rosegrant *et al.*, 1995[22]; Bazza and Ahmad, 2002). In principle, of course, it is true that water-intensive crops become increasingly less profitable relative to less water-using crops if water charges are increased. But in practice, because water costs usually comprise only a small part of farm costs, very high increases in water costs and attendant income reduction are necessary to make these less water-intensive crops more attractive. This is illustrated in Fig. 2.8. Assuming that coefficients are fixed, crop shifts are costless and other costs and prices remain the same, the charge per cubic metre at which crop A (net income 100, water costs of 10 deducted) becomes less profitable than crop B80 (initial net income 80% of crop A, water needs 50% of crop A) is five times the initial charge, while income is slashed by 40%.[23]

Possible 'crops B' will be available to the farmer only where these have a net income comparable to crop A and where water costs are already relatively (very) high. This is rare in practice but occurs in private pressurized irrigation with high fixed costs (Charentes, France: Moynier, 2006), particularly in some groundwater areas (e.g. in Spain, Varela-Ortega, Chapter 14, this volume) where the alternative is rain-fed agriculture.

Of course, a more favourable outcome would be to see farmers adopting higher-value crops instead of lower-value crops. Although such a shift is frequently expected from

[22]'We argue that valuation of water at its opportunity cost will provide incentives for farmers to shift from water-intensive rice to higher-valued, less water-intensive crops after wet-season rice; and in other environments to shift from field crops to fruits and vegetables' (Rosegrant *et al.*, 1995).

[23]For crops B60 and B40 which have initial net income of 60% and 40% of crop A, the increases are even more massive (see Fig. 2.8). Even in the case where water costs represent 30% of the initial net income (a very high value) crop B80 becomes more profitable after multiplying water costs by 2.3, *but* with an unchanged income loss (40%).

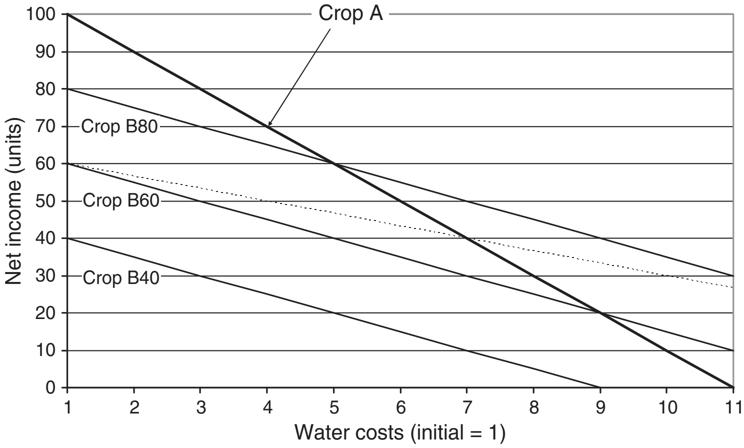

Fig. 2.8. Decrease of crop profitability with water costs.

increased prices, one may wonder in the first place why farmers would have neglected such an opportunity since it was already available to them, and why they would have to wait to see their benefits reduced by higher water costs before adopting it. This will enable us to get a closer scrutiny at farmer decision making regarding crop selection.

It must also be noted that high water use does not always imply low profitability and vice versa. 'Thirsty' crops with high returns include bananas (e.g. Jordan), rice (e.g. Egypt, Iran), sugarcane (parts of India) and qat (Yemen). Lucerne may consume a lot of water but does not have to be low-value, e.g. when in rotation with cereals. Above all, paddy is seldom grown *because* water is free or cheap (Falkenmark and Lundqvist, 1998) but in response to numerous environmental, social and other factors. Crops with lower requirements may not increase farmer incomes (and vice versa) and the impact on water productivity is far from self-evident. When high-value crops are also more water-intensive, higher prices may cause an increase in total demand for water, a phenomenon Dinar and Zilberman (1991) called 'the expansion effect'. In sum, the objectives of farmers (per hactare income), managers (reduce demand) or economists (water productivity) often do not coincide, although policies sometimes posit otherwise.

Economic growth, structural change and urbanization fuel demand for high-value products such as fruits, vegetables and meat (Rao *et al.*, 2004). Although the value of agricultural exports has risen dramatically, cereals continue to occupy more than 50% of the cultivated area worldwide, and fruits, vegetables and related high-value crops are confined to less than 7.5%. No doubt this share will rise but market constraints remain limiting, and cultivation must inevitably be confined to entrepreneurial farmers able to assume the costs and risks of high-return commercial agriculture. Access to groundwater greatly reduces water and related risks, but financial strength, entrepreneurial enterprise and credit access are still all required. Market volatility generates income instability (Hazell *et al.*, 1989; Quiroz and Valdés, 1995; Combes and Guillaumont, 2002) and most poor farmers cannot be expected to incur such risks, even if market volatility can sometimes be moderated by state interventions.

In addition to financial and marketing risk, crop choice is governed by a host of other well-identified factors.[24] These factors

[24]See, for example, Ellis (1998), Pingali and Rosegrant (1995) Quiroz and Valdés (1995), Pingali (2004), Arrojo (2001), Varela-Ortega *et al.* (1998), Dorjee *et al.* (2003); Barghouti *et al.* (2004), Gómez-Limón and Riesgo (2005), Binswanger and Rosenzweig (1986), World Bank (1988).

include: (i) labour constraints; (ii) lack of capital, credit or desire to get indebted; (iii) lack of information on market demand, quality requirements, agricultural techniques and agrochemicals, or adequate skills, etc.; (iv) land tenure uncertainty that hinders investments and adoption of perennial crops; (v) drudgery and health risk; (vi) soil, drainage or climatic constraints; (vii) high marketing costs due to poor transportation means (Delgado, 1995; World Bank, 2005a) and lack of infrastructure (cold storage trucking, refrigeration, etc.) (Barghouti *et al.*, 2004); (viii) the (un)reliability of irrigation supply and possible water quality constraints (Burt and Styles, 1999); and (ix) farmers' strategies, including food security considerations and many ageing farmers with exit strategies and no desire to take risk with new ventures, or to face increased drudgery.

This reminder serves here to dampen the enthusiasm that farm economic problems can be solved by a sweeping shift to high-value, capital-intensive and entrepreneurial agriculture. Another consequence is that farm models that seek to explain crop choice using fixed coefficients and oversimplified decision-making models fail to capture farmer responses, constraints and risks in full, with the implication that modelling approaches probably overstate the mobility of farming systems and their response to prices. Also, the responses are not confined just to farm practices. Farmers bring political pressure to bear when charges are raised and/or may refuse to meet obligations they consider punitive or unfair, break structures, tamper with metres or collude with field staff. Sanctions are difficult – even impossible – to enforce where control at the farm level is so often illusory.

In contrast to water charges, rationing and supply management can be very effective in influencing crop choice. The reasons are perhaps obvious. That water costs are seldom a critical issue does not mean that water is not a critical input. Farmers' indiscipline undermines supply management practices and, faced by shortages, deficit irrigation is a first response. But if schedules and quotas are strictly enforced, farmers perforce have to change their cropping patterns (or equipment) if basic water supplies are insufficient to meet minimum crop water requirements. Besides being a mechanism for managing scarcity and bringing supply and demand into immediate balance, supply management thus impacts on crop choice both in the short and (if sustained year to year) the long term.

Technological Change

By far the most important response to water scarcity has been the tube well revolution. Groundwater accounts for as much as 50% of agricultural value-added under irrigation, with much of it within the boundaries of surface irrigation schemes. Investment in water-saving technologies – buried pipes, sprinklers, micro-irrigation, land-levelling – represents a further response to water scarcity and to consequent high water costs. However, water is not the only factor involved. A profit-maximizing farmer, in principle, invests when (financial) capital and future O&M costs are justified in terms of anticipated increases in net income. Both farmers and conditions vary widely, and the decision to invest in costly equipment is seldom a straightforward response to water conditions but reflects a host of interconnected factors (Caswell and Zilberman, 1985; Green *et al.*, 1996; Schuck and Green, 2001; Moreno and Sunding, 2005). These may include[25]: (i) feasible crops; (ii) environmental conditions (soil quality, slope, plot size and shape, wind, water quality, etc.); (iii) the presence or absence of equipment suppliers and after-sales service; (iv) farmer education, skills, financial capacity and entrepreneurial spirit; (v) the amortization of existing material; and (vi) market opportunities, costs and risks.

[25]For discussion on the adoption of irrigation technology see also de Fraiture and Perry (Chapter 3, this volume), Green and Sunding (1997), Varela-Ortega *et al.* (1998), Dinar and Yaron (1990), Lichtenberg (1989), Sunding (2005), Green *et al.* (1996), Sumpsi Viñas (1998), Molle (2006), Green *et al.* (1996), Scheierling *et al.* (2006b), Dinar and Zilberman (1994), Schuck *et al.* (2005), Skaggs (2001), Shrestha and Gopalakrishnan (1993), Moreno and Sunding (2000).

Moreover, even discounting for risk and associated factors, profit maximization is not always the farmer's major preoccupation. Cropping in Jordan, for instance, can be explained in part by considerations of prestige and leisure (Venot *et al*., Chapter 10, this volume).

Supply management and regulation of water use are sometimes used to dictate farm-level investments in water-saving technologies based on beneficial use or similar grounds. Some governments, supported in many cases by donor agencies, go further and subsidize such investments. Beyond initiating research and pilot schemes, however, such programmes are generally self-defeating, leading to overproduction, accentuated price volatility and discrimination against those who fail to obtain subsidies. Farmers are invariably the best judge of the investments justified in their own circumstances, and governments should limit their role to the regulation of water rights and water use so as to manage conflict, enable reallocation and promote environmental sustainability. Given extensive groundwater capacities, there is in particular little point in subsidizing modern water-saving technologies in massive surface systems which cannot compete with groundwater and which will inevitably remain largely for the production of cereals and other traditional crops.

Pricing, Crops and Technological Change: Empirical Evidence

Agricultural diversification and investments in water-savings technologies often go together, but are driven by market opportunities and total farming conditions rather than by water prices. Broad reviews at national level include that by Yang *et al.* (2003), who conclude that despite strong promotion of agricultural diversification 'the pace of this shift has not accelerated . . . [due to] constraints of marketing channels, processing and transport facilities, and market demand . . . particularly for perishable crops, such as vegetables and fruits'. With market saturation in many markets, they conclude that 'further raising irrigation charges are unlikely to lead to a substantial shift to cash crops'. Siriluck and Kammeier (2003) analysed a nationwide project aimed at fostering agricultural diversification in Thailand. They found that extension and credit packages may encourage some diversification but that 'blueprint' approaches insensitive to household diversity may push farmers into risky ventures and indebtedness. Artificially boosting output of specialty cash crops often sends market prices down, thus reducing the initial benefits of the shift and increasing the risk of bankruptcy.

Case studies provide similar conclusions. Both linear programming at farm and system level, and econometric models have attempted to capture the impact of pricing on cropping patterns and investments. Such models typically assume that farmers are profit-maximizing agents (Pinheiro and Saraiva, 2005), but differ greatly in their treatment of risk and other factors. Price elasticities and other outputs of such models heavily depend on the context, the assumptions made, the variables retained and the adjustments farmers are allowed to make (Ogg and Gollehon, 1989; Scheierling *et al.*, 2004). Most studies are from developed countries (western USA, Israel and southern Europe) and assume volumetric control and water on demand. In Spain, for instance, Varela-Ortega *et al.* (1998) show that to obtain a 10% reduction in water consumption 'irrigators of the Valencia region have to sacrifice up to 70% of their income, compared to 57% of their counterpart in the Castille region and a small 9% in Andalusia'. The low value in Andalusia is explained by the productive potential of this region, its large farms and the availability of alternative crops. Sumpsi Viñas (1998) obtained similar results for the Balbilafuente scheme, concluding that the elasticity of demand depends on farm size, initial water endowments, available crop alternatives and strategies of production (intensive or extensive), all of which differ regionally. Berbel and Gomez-Limón (2000) show for the Guadalquivir and Duero basins that farm incomes have to be decreased by 25% and 49%, respectively, before water demand decreases significantly. These and numerous other studies in Europe (Gómez-Limón and Riesgo, 2004a,b for Spain; Morris *et al.*, 2005 for the UK; Bazzani *et al.*, 2005 and Gallerani *et al.*, 2005 for Italy; Pinheiro and Saraiva, 2005 for Portugal), although undertaken in differing contexts with differing assumptions, hypotheses and coverage, tend to converge on a number of common conclusions:

- Response to price tends to be high for extensive and low for intensive high-value agriculture and depends on the number of crops that can be grown in any given region (which may be limited).
- Water savings due to crop or technological shifts only occur at price levels that severely dent farmers' incomes. If irrigation is extensive or has been developed as a social investment, large subsidies are needed to preserve farming after modernization.
- Water demand under micro-irrigation is inelastic. Once improvements in water-use efficiency have been achieved due to its adoption, further gains are increasingly unlikely.
- Water agency receipts often increase as water prices rise, though this is sometimes more than offset by reductions in water use.
- Because regions, and farmers within regions, are heterogeneous, nationwide policies will not be successful and have negative impacts on those who cannot adjust.

Many of these studies point to the adverse economic and political consequences of raising prices to levels that could impact on cropping and/or technology. Raising water prices sufficiently to impact on use and technology is not only a blunt instrument with widely differing regional impacts, but often results in irrigation becoming unprofitable. The decision on whether to provide subsidies forms part of a wider discussion on agricultural protection – the implication being that quotas are more effective in limiting water use if the concurrent aim is to preserve farm incomes and farming communities.

US studies have more mixed conclusions. While some are in agreement with these conclusions (e.g. Scheierling *et al.*, 2004 for South Platte; Scheierling *et al.*, 2006a,b; Hoyt, 1984; Caswell *et al.*, 1990), others suggest that technological change can occur in response to price (Caswell and Zilberman, 1985; Nieswiadomy, 1985; Negri and Brooks, 1990; Moore *et al.*, 1994). The reasons are unclear but some of the latter US studies appear to fail to establish a satisfactory level of causality between the water price and technological investment (Sunding, 2005), while others do not explore income losses and subsidies sufficiently to be comparable with the European studies. Be that as it may, there are many examples showing that water prices are seldom the primary driver in the adoption of water-saving technology since investment costs are almost invariably far greater than any savings in the water bill. Perry (2001a,b) shows, for central Iran, that the cost of reducing deliveries via such technologies is twice the actual cost of supply by the agency. In Gujarat, tube well farmers have complete flexibility and pay more than 30% of their net income for water, but there is little investment in improved technologies (Cornish *et al.*, 2004). De Fraiture and Perry (Chapter 3, this volume) conclude that 'empirical evidence shows that technology choice is hardly driven by water price' and Varela-Ortega *et al.* (1998) argue that 'the adoption of irrigation technology is not the most significant response to water pricing policies . . . technology adoption in highly productive regions can come about at zero water price rates'. In India (Shah *et al.*, Chapter 9, this volume) or in the Jordan valley (Venot *et al.*, Chapter 10, this volume), micro-irrigation developed when the price was very low, and Sunding (2005) concludes that 'water price is not the most important factor governing irrigation technology adoption' in San Joaquim valley; dissemination of centre pivots in California occurred when water costs were irrelevant (McKnight, 1983).

In practice, investment in water-saving technologies is linked to numerous other interacting factors (Dinar and Zilberman, 1994; Scheierling *et al.*, 2004). Diffusion of drip irrigation in Israel, for instance, was spurred by: (i) higher yields; (ii) subsidies; (iii) sandy soils; and (iv) the reuse of water savings to expand cultivation (Dinar and Zilberman, 1994). In other cases, produce quality (e.g. potatoes in the UK) and reduced labour costs are paramount. Calculations made by Sumpsi Viñas (1998) for vegetable and fruit production in several regions of Spain showed that impacts on yield, quality and labour use make drip and sprinklers more profitable than furrow irrigation. In Hawaii, drip irrigation was widespread in sugarcane because it increased yields, saved labour (and some water) and allowed expansion of cultivation on marginal and sandy soils (Shrestha and Gopalakrishnan, 1993). In Tunisia, although modernization targeted water saving, on-farm water use was not significantly altered, though higher yields and incomes were obtained (Al-Atiri *et al.*, 2004). García Mollá's (2000) study of Valencia in Spain and Carles *et al.*'s (1999) review of nine irrigation schemes also demonstrated that adoption of drip irrigation was motivated by reduced labour, enhanced quality, convenience and fertilizer saving.

Finally, contrary to common wisdom, the use of water-saving technology at the farm level does not necessarily mean that the fraction of applied water that is depleted (actually transpired or evaporated to the atmosphere) has been reduced. Soil evaporation is often reduced but crop evaporation is generally increased because of better and timelier application (Burt *et al.*, 2001; Perry, 2001a,b). Furthermore, evidence from arid and semi-arid regions, and more generally if land is not a limiting factor, suggests that water savings, to the extent they are obtained, are generally retained by the farmer or his neighbours to expand the cropped area. While benefits accrue to those expanding this area, the fraction of water depleted typically rises and return flows and aquifer recharge decline. García Mollá's (2000) study in Valencia revealed that districts adopting drip irrigation have attempted to maximize the area under cultivation. Similar situations have been described in countries such as Tunisia (Feuillette, 2001), India (Moench *et al.*, 2003), Spain (Carles *et al.*, 1999), Israel (Dinar and Zilberman, 1994), Morocco, the USA (Caswell, 1998; Huffaker *et al.*, 2000; Skaggs, 2001; Aillery and Gollehon, 2003; Huffaker and Whittlesey, 2003) and Hawaii (Shrestha and Gopalakrishnan, 1993). Public subsidies[26] aimed at improving efficiencies and releasing water for other uses are thus often counterproductive.

In sum, adoption of water-saving technology is seldom driven by water scarcity or water prices, but by an association of benefits that play out together: yield increases allowed by better and more homogeneous application of water, better quality and a more homogeneous product, bringing substantial increases in the market price, better application of fertilizers and chemicals, decreased labour costs, decrease in return flows contributing to reducing the leaching of fertilizer and pesticides and to controlling soil erosion are some of the associated benefits.[27] Further incentives are clearly linked to the possibility of using water savings to expand cultivation where land is not a constraint, and to that of capitalizing on existing pressurized supply when water is pumped from wells (Caswell and Zilberman, 1985; García Mollá, 2000; Becker and Lavee,

[26]Many countries subsidize micro-irrigation and farm-level improvement. In Morocco, for example, they are subsidized at a level of 30–40% and farmers are granted bonuses (Belghiti, 2005a) because technologies are too costly for farmers, but even then adoption is slow (Tizaoui, 2004). In Israel, micro-irrigation is generalized but the growth of 700% observed during 1975–1982 was spurred by heavy government subsidies that made the shift profitable (Shevah and Kohen, 1997). In the USA, the conservation of groundwater and surface water has been promoted by the Environmental Quality Incentives Program initiated in 1997, whereby cost-sharing may pay up to 75% of the costs of eligible conservation practices (Scheierling *et al.*, 2006a).

[27]For further discussion, refer to Caswell and Zilberman (1985, 1990), Dinar and Zilberman (1991, 1994), Caswell (1998), Morris *et al.* (2005), Wierenga and Hendrickx (1985), Carles *et al.* (1999), Skaggs (2001), Sumpsi Viñas (1998), McKnight (1983), Scheierling *et al.* (2006a), Becker and Lavee (2002) and García Mollá (2000).

2002). As a rule, these shifts generally result more from changes in market opportunities, output prices and subsidies (e.g. the Common Agricultural Policy (CAP) in Europe) than from changes in input costs.

PRICING AS AN ECONOMIC INSTRUMENT: ALLOCATION BETWEEN SECTORS[28]

Introduction

Urban growth and industrialization fuel rising water demands. According to the World Bank Strategy of 1993 'setting prices at the right level is not enough; prices need to be paid if they are to enhance the efficient allocation of resources' (World Bank, 1993); for Johansson (2000): 'The fundamental role of prices is to help allocate scarce resources among competing uses and users. One way to achieve an efficient allocation of water is to price its consumption correctly.' With higher prices that reflect opportunity cost, the reasoning goes, low-value activities are phased out, thus releasing water for high-value uses and raising social welfare.

As water shifts, *allocation stress*[29] moderates and economic gains are realized (Dinar, 1998; Rosegrant and Cline, 2002; Merrett, 2003; Hansen and Bhatia, 2004): 'supporting 100,000 high-tech California jobs requires some 250 million gallons of water a year; the same amount of water used in the agricultural sector sustains fewer than 10 jobs, a stunning difference' (Gleick, 2000). Elsewhere Gleick says: 'as much as half of all water diverted for agriculture never yields any food. Thus even modest improvements in agricultural efficiency could free up huge quantities of water.' But these and similar statements[30] need to be challenged. It is true that irrigation consumes much more water than urban uses, both absolutely and relative to diversions, but this is inherent to the activity (Abernethy, 2005) and it does not follow that increased 'agricultural efficiency' is a precondition for meeting other needs. To recapitulate:

- Irrigation may use uncontrolled and other marginal sources that may be unable to provide the security and quality needed by domestic or industrial users (Savenije and van der Zaag, 2002).
- There may be no hydraulic connectivity between irrigation and potential urban uses, and transfers and storage may be impracticable or prohibitively expensive (Smith *et al.*, 1997).
- Basin efficiencies are much higher than subsystem efficiencies (Frederiksen, 1996; Keller *et al.*, 1996; Perry, 1999; Molle *et al.*, 2004).
- Response to scarcity means that farmers use water more efficiently than is commonly assumed, adopting conservation measures and conjunctive use that offset the impact of reduced supply.

Moreover, if reallocation of water becomes necessary and is feasible, this almost invariably occurs, though not necessarily at lowest cost or in the most sustainable manner. Deficiencies in urban systems are thus primarily due to financial constraints and political priorities, and not to water being 'locked up' in 'inefficient' irrigation. The following subsections review these issues further under three headings: (i) allocation or financial stress?; (ii) transfer mechanisms; and (iii) implications. Issues associated with environmental externalities are discussed in the next section.

[28]This section is largely derived from Molle and Berkoff (2006), to which the reader is referred for further details.

[29]The allocation stress is typified by Bate (2002): 'The effect of under-priced water is that farmers use inefficient irrigation technologies to produce uneconomic goods at the expense of lucrative alternative economic activities.' The opportunity costs of this misallocation can be vast. See also Dinar and Subramanian (1997).

[30]See similar statements in Winpenny (1997), Simon (1998), IRN (2003), Postel (2001), Hansen and Bhatia (2004), ESCWA (1999) and Colby (1990), among others.

Allocation or Financial Stress?

Allocation stress

Allocation stress is said to occur when high-value sectors are deprived of water that is locked into lower-value activities. But the existence of a significant allocation gap is doubtful. In practice, farmers are 'losing out' (Winpenny, 1994), urban interests get the 'upper hand' (Lundqvist, 1993) and 'cities will continue to siphon water away from agriculture' (Postel, 1999). Transfers out of agriculture or ecological reserves (to the extent necessary and feasible) may be minor or major, gradual or outright, surreptitious or open, on the surface or underground, and with or without compensation, but by and large cities procure the water they need (Molle and Berkoff, 2006), in both the shorter and longer terms.

Priority in a drought is almost invariably given to urban uses, and to industry and services in particular. For example, shortages in industry and tourism in the 'Eastern Seaboard' near Bangkok have been quickly diffused by the implementation of six inter-basin transfers and drilling of 290 artesian wells for short-term relief (Samabuddhi, 2005).[31] Page (2001) cites a survey of the Hebei province that showed 'how local officials enforced restrictions on farmers but overlooked those on industry to lure projects from which they could profit'. Amman's supply was hardly impacted by the 2000/01 drought; the California State Water Project cut-off farmers in 1991, and the Bureau of Reclamation reduced supplies in the Central Valley by 75% (Anderson and Snyder, 1997); Jakarta's golf courses were supplied in the major 1994 drought; and in Cyprus farm supplies were cut by 50% in a 3-year drought but supplies to the 2 million tourists were maintained (Barlow and Clarke, 2003). Other examples where agriculture suffered first include Chennai, India (Ramakrishnan, 2002), the Guadaquiver basin in Spain (Fereres and Cena, 1997), the Alentejo region in Portugal (Caldas *et al*., 1997) and Manila (McIntosh, 2003).

Whether longer-term investments in services and industry are constrained by water remains perhaps a matter of debate. Very high water-consuming industries, such as aluminium, are unlikely to settle in water-short areas, and suggestions have been made that water-intensive industries should be moved, e.g. inland from coastal China (Chan and Shimou, 1999). Many cities appear to be in the *wrong* place (Winpenny, 1994) and have to opt for more distant and costly transfers after exhausting nearby water supplies. But they can still continue to grow rapidly: Chennai, Mexico City, Las Vegas, Tianjin and Amman are widely differing cities that all illustrate this despite their very limited nearby resources. Ta'iz grew by 7.9% between 1986 and 1994, despite being one of the most water-stressed cities in the world. Even in water-abundant areas, cities outstrip proximate resources when located in upper catchments (e.g. São Paulo, Atlanta, Kuala Lumpur) or in small coastal catchments (e.g. Manila, New York, Boston). Although the costs of water vary greatly depending on local circumstances, there is little evidence that water constraints seriously impact on urban growth; and when this is the case it is rarely due to water being locked up in agriculture, except in situations where formal water rights may dictate so (e.g. western USA).

Financial and political stress

That cities, by and large, are able to obtain the water they need does not, of course, mean that water supply and sanitation (WSS) services have no deficiencies. Far from it. But these deficiencies reflect political priorities and financial constraints rather than water availability as such. In Europe for instance, in historic times, extension of WSS facilities beyond the affluent

[31] The Finance Minister is reported to have told senior bureaucrats that their 'heads are pledged as a guarantee, since this issue is a problem for the entire country . . . I don't want to hear again that industries along the Eastern Seaboard are facing water problems, whether it's this year or in any other year'.

can be attributed to a combination of the hygienist movement, a perceived 'threat from below' (Chaplin, 1999) and/or the need 'to preserve order, cleanliness and a healthy workforce' (Goubert, 1986). As early as the mid-18th century it was recognized that 'prevention of further environmental degradation was cheaper and more effective . . . than continuing with expenditure on poor relief' (Chaplin, 1999). Elites in Guayaquil (Swyngedouw, 2003) and Monterrey (Bennett, 1995) reacted in more recent times to social unrest. In contrast, Chaplin (1999) attributes the negative picture in India to a failure by the upper classes to pressure the government to invest. WSS investments differ in their political rewards and the key question is 'who will pay?' rather than 'where is the water?'.

Political considerations are compounded by financial and institutional constraints. Few cities in developing countries have been able to keep pace with inward migration (Lundqvist *et al.*, 2003) and the costs of collecting, conveying and disposing of water in line with city expansion have proven beyond their financial capacity. This has generally remained true throughout their history, when the population was far lower than now just as much as once the mega-cities of the present day had developed. Even in water-abundant regions, developing country cities have deficient WSS systems (e.g. Lagos, Dhaka, and Ho Chi Minh City). 'The root cause [of poor water supply to population] is our negligence and our resignation in the face of inequality' (Camdessus and Winpenny, 2003). Other documents addressing this issue similarly fail to refer to physical scarcity as a constraint (Anton, 1995; UNESCO, 2003). The question of 'who will pay' is key to understanding WSS conditions in cities. Capital cities are particularly well placed to access public funds (e.g. Mexico: Connolly, 1999) and how taxes are shared between local bodies, and state and federal governments, has an important bearing on the outcome. Some cities attract foreign subsidies (e.g. EU funds for Athens) or benefit from geopolitical considerations (e.g. Amman) or broad reconstruction factors (e.g. Phnom Penh). If society is receptive to privatization, the financial burden can be shifted to users, as in the UK, but elsewhere privatization and public–private partnerships have had mixed results in view of the risks, poor financial returns and political sensitivities (SIWI, 2004).

By and large, cities can secure necessary water resources. The mechanisms adopted to achieve the transfer, however, vary greatly. They depend, in particular, on the characteristics of the hydrological system, the nature and practice of government and on the strength of the regulatory and water rights systems. They are discussed below under three headings: expropriation (with and without compensation), opportunity cost pricing and markets.

Reallocation: Bureaucratic Expropriation, Administered Prices and Markets

Expropriation

An extensive literature review suggests that governments, urban utilities and industries commonly reallocate water by bureaucratic action (Molle and Berkoff, 2006). When successive urban projects take amounts that are small relative to river flows, reallocation can occur by stealth, with the impact on downstream farmers and ecosystems obscured by natural hydrologic variability. Even more prevalent than such reallocation of surface flows is the 'hidden' expropriation of groundwater resources as urban users deepen wells and increase pumping: approximately 1.5–2.0 billion people are said to rely on groundwater for domestic consumption, including 1 billion urban inhabitants in Asia (Foster, 1999), and industries often access groundwater directly because it is secure and needs no treatment. Where confiscation by stealth is impracticable, utilities may exercise *force majeure* – supported by politicians – and deprive farmers and other users outright. Since property rights are seldom clearly demarcated, confiscation may be legal in the sense that governments usually retain the final say on who receives water in the national interest. A further argument used to rationalize direct confiscation is that irrigation was a (heavily subsidized) gift of government in the first place. In cases where formal rights are effective, expropriation is precluded in the absence of financial compensation.

Expropriation is, in its nature, inequitable, depriving farmers of their traditional livelihood without recourse, accelerating the process of structural change and aggravating income inequities. Thus, although it is conceptually the simplest mechanism for effecting water transfers, direct expropriation can be problematic for any government, even an authoritarian one, especially in contexts where the local economy revolves around irrigated agriculture. This has led governments to consider compensation schemes on a case-by-case basis, even where formal property rights do not exist. This can take the form of *either* complementary action to ensure that the impact on irrigation is minimized *or* financial compensation for the losses incurred.

An example of complementary action was by El Paso which obtained water from the Rio Grande on condition that it reduced per capita consumption, recycled sewage water and eliminated leakage (Earl, 1996). Dongyang city obtained water from a dam managed by the Yiwu city, but had to finance an increase in the height of the dam and line irrigation canals (Liu, 2003). The 1998 agreement between the Imperial Valley Irrigation district and the Southern California Metropolitan Water Authority (MWA) included the lining of the All-American Canal by MWA with usufruct rights to the 100 Mm^3 thought to be 'conserved' passed to Southern California metropolitan area (Cortez-Lara and Garcia-Acevedo, 2000); similarly, the Upper Ganga canal was lined so that 'seepage losses' could be reallocated to Delhi. In both cases, however, these transfers were in practice at the expense of downstream groundwater users, who in the Californian case were Mexican farmers. Molle *et al.* (2004) use an example from Central Iran to show that in 'closed basins', where most or all resources are committed (often overcommitted), conservation measures do not save water, but merely reallocate it across the basin in a way that is not always perceptible.

Examples of compensation for water transfers include the buying out of agricultural wells around some cities (e.g. in Phoenix or Chennai); the diversion of water from neighbouring irrigation reservoirs to serve cities (e.g. Tsingtao in China where irrigation reservoirs were converted to urban use in preference to paying higher rates for Yellow River water); and the purchase of reservoir storage for hydro-generation from farmers during droughts in the Guadalquivir River basin, Spain. The merit of these and similar arrangements is that the transfer between irrigation and the utility can be adapted to specific local realities to the benefit of both sides. The government ultimately acts as mediator between the two and as the guarantor that the agreement will be honoured.

Opportunity cost pricing

Rather than expropriate water – with or without compensation – transfers can, in principle, be forced by full economic pricing of supply.[32] The World Bank's 1993 water policy and repetition by resource economists has disseminated the idea of the need for reallocation from low- to high-value uses, and this idea has been incorporated in national policy and legal documents. Zimbabwe's 1994 Irrigation Policy and Strategy, for example, states: 'Since water is scarce, its opportunity cost should be taken into consideration in determining price' (Nyoni, 1999). Despite these intentions and policies, however, charging economic prices

[32]While some see this as a desirable or compelling objective (although some phasing might be necessary to get there) (Khanna and Sheng, 2000; Rosegrant *et al.*, 1995; EU, 2000a; GWP-TAC, 2000; Plaut, 2000; Socratous, 2000; Saleth, 2001; Ünver and Gupta, 2003), others admit that it might be a far-fetched – or impractical – objective, especially when not even O&M costs are recovered) (Sampath, 1992; Smith *et al.*, 1997; Thobani, 1997; Asad *et al.*, 1999; Garrido, 2002; World Bank, 2003b).

has in practice remained elusive (Bosworth *et al.*, 2002; Kulshreshtha, 2002; ICID, 2004). Acknowledging the 'yawning gap between simple economic principles . . . and on-the-ground reality' that has prevailed for decades, the World Bank (2003) reconsidered the issue and singled out two main reasons for this gap: first, the impossibility 'to explain to the general public (let alone to angry farmers) why they should pay for something that doesn't cost anything to produce'; and second, the fact that 'those who have implicit or explicit rights to use of the resource consider (appropriately) such proposals to be the confiscation of property' (see Molle and Berkoff, Chapter 1, this volume).

A further reason why economic pricing is impractical (Asad *et al.*, 1999) and has seldom if ever been adopted (ICID, 2004) is that opportunity costs are location- and time-specific, and operate at the margin, falling off drastically once effective urban demand at any specific location has been satisfied (Savenije and van der Zaag, 2002). Moreover, the opportunity cost price does not equal the full opportunity value in urban uses but an intermediate value determined by the shape of the relevant demand curves given that a fixed amount of water must be allocated between competing uses when externality and other costs vary (Green, 2003). Even if this price could, in practice, be estimated, the implication is that high charges would be paid by those in irrigation schemes in direct competition with neighbouring urban areas, and that those further away and not in competition would pay much lower prices. As noted earlier, charging for opportunity costs would also be politically and socially self-defeating since the order of magnitude of these costs would bankrupt most of the irrigation activities affected (Bate, 2002; Tardieu and Préfol, 2002; The Economist, 2003[33]), especially when irrigation is inherently uneconomic (first section). Despite these impediments, two countervailing arguments are sometimes asserted:

- Stripped of normative content with regard to price fixing, the estimation of opportunity values in alternative uses sheds light on how much is recovered from users, paid by the state and left uncovered. This is a central argument of the EU's Water Framework Directive.
- Even if full opportunity cost pricing is impracticable, moving towards higher water charges might still instil a degree of market logic, promote structural shifts in the rural community, and favour those who can make the best use of available irrigation supplies.

Charging opportunity costs is nevertheless comparable to expropriation in that those who lose their water as a result of an inability to pay receive no compensation (Cummings and Nercessiantz, 1992) and this can be perceived as expropriation by those who have customary rights or who have bought land with the value of water incorporated in the price (Rosegrant and Binswanger, 1994; Garrido, 1999; World Bank, 2003a,b). Given also the potential for inefficiency and rent-seeking in the context of bureaucratic involvement, many point to water markets as a preferable solution to either expropriation or opportunity cost pricing to resolve allocation problems (Thobani, 1997; Bate, 2002).

Market reallocation

Small-scale water markets have long existed. The ancient markets of Alicante are well known (Maass and Anderson, 1978). More generally, community-based irrigation supplied by springs or qanats (Beaumont *et al.*, 1989) often has well-defined individual rights that lend themselves to temporary or permanent transactions. Most occur in 'spot markets': neighbours swap, lend, borrow, sell or buy water turns in order to fine-tune supply to individual demands. This also

[33] *The Economist* (2003) emphasizes that it is not 'politically plausible to suggest that farmers must always pay the full costs of their water. Water for irrigation is highly price-inelastic: since farmers have little alternative but to use the stuff, charging the full cost could simply drive them into bankruptcy'.

occurs in large-scale irrigation systems if supply is sufficiently defined in terms of time or discharge to permit quantitative estimation (a notable example being the warabandi systems of Pakistan and north-west India). Recently, groundwater markets have spread in South Asia and elsewhere although these are perhaps more akin to buying of a service than of the water itself (Shah, 1993). At these scales, transaction costs are minimized because users know each other (Reidinger, 1994), can readily communicate, and transfers are across short distances without costly infrastructure or significant losses. Permanent transfer of ownership is also socially controlled and local third-party impacts are easily identified.

Traditional markets reallocated water primarily within agriculture, although conversion of wells to water supply for tanker markets also occurs (e.g. in Jordan and India). Market reallocation has also sometimes performed well at a larger scale when the institutional conditions allow. Examples include trading of Rio Grande water in Texas (Chang and Griffin, 1992), the Westlands Water District in California (Brozovic *et al.*, 2002) and the Colorado-Big-Thompson scheme (Howe, 1986; Mariño and Kemper, 1999), where most transactions are spot transactions and rental (Carey and Sunding, 2001), but also include permanent transfers from agriculture to other sectors (Howe and Goemans, 2003). In South Africa's Orange River basin, trading has occurred between commercial farms (Backeberg, 2006). In Australia, transfers within and among distant irrigated areas have developed in the last 10 years (90% being temporary transfers) (Isaac, 2002; Turral *et al.*, 2004). Bauer's (2004) review of the Chilean experience describes active markets in the Limari basin (mostly short-term reallocation between irrigators supplied by the same reservoir), and in the Maipo and Mapocho basins close to Santiago (4% of all water rights were traded between 1990 and 1997, half being acquired by municipal utilities: Alicera *et al.*, 1999). In Mexico, trading occurs within large irrigation schemes, but interstate transfers are closely regulated (Simpson and Ringskog, 1997).

As the scale and number of users increase, however, water's well-known characteristics (see first section) make it prone to market failure (Livingston, 1995). Defining property rights can be very difficult; economies of scale invite natural monopolies (Easter and Feder, 1998); and the transaction costs associated with markets – information, regulation and enforcement – are typically large. Above all, third-party and externality effects are pervasive, and it is often very difficult to link particular flows with particular uses or users. Markets in the USA have, for instance, been constrained by the lengthy and costly litigation to which third-party impacts often give rise (Dellapenna, 2000; Kenney, 2003; Libecap, 2003). Market transactions within the Colorado-Big-Thompson system may work well, but this is partly because they are confined within one water district that holds the right to all return flows (Howe and Goemans, 2003; Libecap, 2003). China suspended an experiment in interprovincial trading once the return flow and environmental impacts became evident (Fu and Hu, 2002).

Moreover, water markets fail to account for scheme- and regional-level impacts of transfers. The transfer of some water rights to non-agricultural investors attached to acequias in New Mexico, for example, weakened management and maintenance of the system as a whole (Klein-Robbenhaar, 1996). Frederick (1998) reports that 'when farmers want to sell water to cities, irrigation districts resist, fearing the loss of agricultural jobs', while Wahl (1993) acknowledges that 'most agricultural water districts have viewed the potential for water transfers only very tentatively out of concern over the security of their water rights and potentially adverse effects on the districts and local communities'. The severity of impacts on the area of origin varies greatly (Gopalakrishnan, 1973; Charney and Woodward, 1990; Howe *et al.*, 1990). Sunk costs in social and non-irrigation economic infrastructure, for instance, may be a strong argument for preserving irrigation, but cannot be reflected in a market price.

Finally, markets may open the door for opportunistic and monopolistic behaviour. Bjornlund and McKay (1999) observed that in Australia, opportunistic buyers were able to exert undue pressure on sellers to obtain lower prices. Bauer (1997) and Hadjigeorgalis (1999) showed that in Chile, 'many small farmers are liquidity-constrained and often have sold rights to pay off large debts'; as 'land is of little value without water . . . it is not expected to observe farmers selling water rights unless they were exiting agriculture or facing liquidity constraint'. In Australia, on the other hand, 57% of water permanently traded was due to farmers having excess water or reducing their irrigation areas (Turral *et al.*, 2004). In California, presumably, transfers between large commercial farms reflect mere shifts in economic opportunities.

Although attractive in principle, the complexity of establishing markets for tradable water rights is formidable (CEPAL, 1995; Livingston, 1995; Siamwalla and Roche, 2001). Positive experience is confined to countries (e.g. the USA, Australia and Chile) having a sound knowledge of hydrology; a comprehensive and modern hydraulic infrastructure (notably of storage); strong legal, institutional and regulatory backgrounds; and relatively wealthy stakeholders. Proposals for the adoption of markets in tradable rights in countries where hydrologic data are scarce, physical infrastructure is lacking, water rights are ill-defined, farmers are numerous and small, and states have generally weak and ill-developed monitoring and enforcement capacity are unrealistic for the foreseeable future (see, e.g., Tanzania in van Koppen *et al.*, Chapter 6, this volume).

Implications

Differences between administrative and market allocation are not perhaps as large as sometimes stated (Mariño and Kemper, 1999). They both require considerable knowledge of the hydrology, control of the water regime, a command over who uses what water where and when and mechanisms for enforcement and dispute resolution. Differences in the effectiveness of regulatory structures may well reveal cultural or ideological values – even local idiosyncrasies (e.g. preference for licenses in Japan or France: Tardieu and Préfol, 2002 or market mechanisms in Chile), rather than degrees of efficacy.

Differences of opinion nevertheless persist between those who emphasize government failure and those who emphasize market failure. The former view state bureaucracies as at best inefficient and at worst subject to corruption and rent-seeking (Rosegrant and Binswanger, 1994; Holden and Thobani, 1996; Thobani, 1997; Easter *et al.*, 1999) and – in the USA – consider that public welfare and public trust doctrines destroy private property and hinder transfers towards higher value uses (Anderson and Snyder, 1997; Gardner, 2003). However, the majority of observers are doubtful that markets can constitute a major tool for the reallocation of water, no matter how theoretically desirable they may be, most especially in developing countries (Colby, 1990; CEPAL, 1995; Livingston, 1995; Morris, 1996; Gaffney, 1997; Frederick, 1998; McNeill, 1998; Dellapenna, 2000; Meinzen-Dick and Appasamy, 2002; Libecap, 2003; Kenney, 2006; Solanes and Jouravlev, 2006).

Markets can no doubt be facilitated at community and local level (Brown, 1997), but water allocation at higher levels requires a 'delicate interplay' between administrative and market control. This 'delicate interplay' would perhaps be best served by a more systematic adoption of compensation arrangements that recognize the economic benefits from reallocation – and the fact that urban interests will obtain their water needs – and also ensure transparency and that the interests of those deprived are taken into account. Ideally, the urban utility and the affected farmers would negotiate face to face, with both in effect faced by the opportunity cost of the water in dispute. The government regulator would, in principle, act as moderator and guarantor, and intervene more generally to safeguard farmers' interests and ensure that environmental externalities and third-party effects are taken into account. No doubt such a system would be open to abuse (government failure would not be abolished), but as regulation strengthens, negotiated compensation could increasingly approximate to regulated markets in which the particular circumstances of the water in dispute are taken into account.

PRICING AS AN ENVIRONMENTAL INSTRUMENT: WATER QUALITY AND SUSTAINABILITY

Introduction

So long as diversions are small relative to the water resource, consumptive and in-stream users are unconstrained in what they do and most water is left to the natural environment as the default *user of last resort* (see first section). But as diversions increase, especially for agriculture, and as in-stream users (e.g. hydroelectric dams) alter flow regimes, wetlands and deltas dry up, water tables and base flows decline, the natural ecology suffers and pollution is concentrated in the limited flow that remains. As a river basin closes, therefore, action must be taken to limit diversions if environmental flows and values are to be protected. What remains is typically diverted by irrigation, and agriculture rather than the environment becomes *the residual user*.

Both agriculture and urban uses contribute directly to pollution of streams and aquifers, sometimes making water unusable for domestic use. Direct agricultural pollution in the USA is said to be $9 billion per year (Bate, 2002). Despite 13 rivers flowing through the city, the degradation of their water due to agricultural and M&I uses has forced Jakarta to tap surface sources 78 km away (McIntosh, 2003); a similar situation is found in Seville because of pesticide and fertilizer residues in the Guadalquivir river; in Chinese cities (Bhatia and Falkenmark, 1993), including Chengdu, where water pollution and silt have forced the closure of two river intakes and the government is investing heavily in watershed rehabilitation (McIntosh, 2003). Irrigation is also responsible for waterlogging and soil salinization as water is diverted to poorly drained low-lying lands within, and at the tail of, irrigation schemes. Other externalities include the mobilization of silt due to catchment changes, which can have devastating impacts on river morphology (famously for the Yellow River), and the mobilization of toxic elements from the soil by leaching. Drainage of the Plain of Reeds in the Mekong delta, for example, releases acidity in waterways, while selenium in California has provoked high mortality of wild fowl in receiving wetlands (Wichelns, 2003).

With regard to groundwater, springs and wetlands fed by groundwater dry up in response to falling water tables (e.g. Azraq aquifer in Jordan) and base flows in rivers decline; falling water yields and water tables lead to higher pumping costs and to the expropriation of poorer farmers and others unable to afford ever-deeper wells (Kendy *et al.*, 2003 for China): falling water tables also aggravate salinity intrusion in coastal aquifers; especially in urban areas, land subsidence reduces aquifer storage and adversely impacts on infrastructure (Nair, 1991); and declining quality due to direct agricultural pollution compounds that from domestic use, industry and landfills (Sampat, 2000).

Environmentalists have vested high hopes in pricing mechanisms as a means of reducing excessive abstraction of water from ecosystems and of decreasing environmental degradation (de Moor and Calami, 1997; Avis *et al.*, 2000). Hodge and Adams (1997) argue that 'the price [of water] could be raised until the level of demand was consistent with the environmental constraints on supply'. Nevertheless, though there is an enormous amount of literature on valuing the environment, there has been limited work on how these values can be incorporated in irrigation pricing and few practical examples of where this has been attempted. As in the case of opportunity cost pricing (previous section), there appears to be little agreement as to how this should be done, and not much hope that farmers would have much understanding of why they should pay such costs. The discussion in this subsection is therefore relatively brief, reflecting as it does the limited evidence in the literature.

Environmental Pricing Mechanisms

The user-pays and polluter-pays principles embody the idea that quantity and quality externalities should be reflected in the price paid by water users as an incentive to reduce adverse environmental impacts and the emission of pollutants. These principles are much more forcefully applied in M&I (given the relative simplicity of volumetric charging and point-source pollution control) than in agriculture, given the problems of volumetric control in irrigation and the intractability of controlling and monitoring diffuse pollution from fertilizers and pesticides (UNEP, 2000).

The EU's Water Framework Directive goes some way in the direction of introducing environmental pricing in agriculture when it states that water charges should 'act as an incentive for the sustainable use of water resources and to recover the costs of water services by economic sector' (EU, 2000b) rather than be adopted for allocation purposes. Nevertheless, both full cost recovery and internalization of environmental externalities are widely seen as ambitious objectives and are, in many cases, impracticable. Modelling, for instance, suggests that much of Mediterranean irrigated agriculture would be jeopardized by strict application of the Directive (Berbel *et al.*, 2005). Mechanisms that have been suggested for irrigation pricing include both negative and positive incentives:

- *Resource charges*. Imposing a resource charge on irrigation equivalent to net externality costs has been suggested to limit diversions and protect the environment. Such charges, in principle, would be imposed on the scheme and passed down to the farmer as a component of the irrigation charge. In practice, however, charging even for recurrent O&M is difficult (as shown earlier) and resource charges have seldom been more than a small administrative fee aiming to recover the costs of resource management (in China, the UK, Spain, Peru, etc.). As far as is known, they have never been high enough to impact on irrigation diversions. Groundwater abstraction fees could, in theory, also be levied on a volumetric basis to limit abstractions to recharge or to some other defined sustainable level. In practice, however, they degenerate into a flat tax, and collection of volumetric charges remains an insurmountable issue, at least in developing countries (Albiac *et al.*, 2006).
- *Pollution charges*. Pollution charges are an incentive for reducing water use and pollutant discharge, though few countries have applied them in irrigation. Denmark is an exception where farmers are subject to the 1994 'Green Tax Reform' that imposes a water rate of €0.55/m^3 of raw water extracted. Further environmental fees are likely given concerns over pesticide contamination of groundwater. Green taxes also exist in Sweden, the UK, the Netherlands, Germany and Croatia (Berbel *et al.*, Chapter 13, this volume; Wright and Mallia, 2003). In France, farmers pay pollution fees for water used in cattle husbandry, but not in crop production. Income from such charges generally goes to the government budget rather than being used to resolve pollution issues, and are seldom high enough to alter behaviour significantly (Young, 1994).
- *Treatment or remediation charges*. Pollution charges may be more acceptable to farmers if used for remedial works within the scheme or in irrigation more widely – thus 'internalizing externalities' – for instance, to help resolve waterlogging, salinity and other problems that impact on scheme production. In South Australia, the government covers the costs of salinity management caused by irrigation projects constructed before 1988, but environmental externalities are charged for all subsequent projects in a two-part price structure. The environmental part of the charge is used to cover the cost of renovation or construction of infrastructure needed to reduce water quality-related externalities (Easter and Liu, 2005).

- *Taxes and rebates*. Rather than specific charges, pollution abatement programmes are more generally met through general taxes. These may, however, be limited to taxes on water users, introducing a degree of cross-subsidization, with the money collected used to treat the wastewater generated not only by the user but also by other dischargers, be they cities, cattle farmers or industries (as in the Basin Agencies in France). In Korea, in some upper catchments, pesticide and fertilizer use has been prohibited with 25% of the funds generated from domestic consumers along the river used as 'income compensation' for upstream farmers who suffer financial loss due to these environmental regulations (Min, 2004). Rather than being taxed, farmers may receive a tax rebate. In western Canada, for instance, rural municipalities have used the municipal tax system as a tool for encouraging specific behaviour by producers. They offered rebates to landowners who implement environmental practices on their land (e.g. grazing land) (Fairley, 1997).
- *Subsidies*. 'Delinking' farm subsidies from direct production payments under the EU reforms (Berbel *et al*., Chapter 13, this volume) is a major attempt to build on existing programmes that have 'paid' farmers to adopt environmentally sustainable practices. Comparable payments are made directly to farmers in Switzerland who participate in three main ecological programmes: integrated production, organic farming and ecological compensation (extensive use of meadows). By 1996, 60% of agricultural area in Switzerland was farmed based on integrated production methods and 5% of the area met organic farming standards. The loss of income is said to be less than if the same effect had to be met through product price increases (Pfefferli and Zimmermann, 1997). In Germany, revenue from water taxes is often used to compensate farmers for restrictions on fertilizer use in vulnerable areas. This idea is also behind the wave of payments for 'environmental services', at the catchment level, for example.
- *Pollution permits*. Pollution permits for nitrogen or another pollutant are akin to quotas for water use. Restrictions on farm animal numbers are used in Europe as a proxy for pollution permits, e.g. in the Netherlands where the primary objective has been to limit groundwater contamination from pig and other intensive operations. As in the case of water quotas, 'permissions to pollute' are often more easily administered and have less implication in terms of welfare losses than a comparable tax on nitrogen utilization or on water use (Martínez and Albiac, 2004, 2006). Effluent permits can also, in principle, be made tradable although this is rare in agriculture. A programme in California with regard to selenium has been successful (Young and Karkoski, 2000) and, although comparable trading regimes have yet to be applied to irrigation or farming in Europe, they are being increasingly adopted in other sectors.

Water Pricing as an Environmental Instrument

Several conclusions can be drawn from this short review. Price incentives for the preservation and restoration of environmental sustainability and water quality have mostly been adopted in the non-agricultural sectors and generally in developed countries. While there have been major programmes that aim, for instance, to restore wetlands or tackle waterlogging and salinization in developing countries, these have almost invariably been funded by government and donors and pricing has seldom, if ever, been significant in controlling these ill-effects. With respect to nutrients and pesticide pollution, their diffuse nature makes them very difficult to measure and control, even in developed countries.

There are a variety of potential pricing schemes ranging from the straightforward application of the user-pays and polluter-pays principles, through partial or full cross-subsidizing by other water users, to full state subsidies. Implementation of the user-pays principle is constrained by all the issues related to irrigation charges discussed in earlier subsections, though any charge that limits water use should have some positive environmental impact. However, the feasibility of major additional environmental charges must be doubted. With regard to pollution, potential interventions are numerous although again problematic in developing countries. They vary from individual prevention incentives (stop the polluting activity) to individual remediation (do it better: use organic farming, extensive pastures, keep cattle sludge in farm reservoirs), to individual treatment (clean up your mess before releasing it), to collective treatment (state infrastructure funded by taxes on water users or the public).

Experience in developing countries suggests that negative incentives, though often feasible in the domestic and industrial sectors (where costs can be internalized within utilities and industrial firms), are often replaced by positive incentives in the agriculture sector whereby the polluter is subsidized to improve his environmental management: subsidies address either the cost of doing so, or the foregone benefits from abandoning polluting (but productive) practices. Payment for watershed services, again, is a good example of a positive incentive. Likewise, Varela-Ortega (Chapter 14, this volume) showed that among the various policies implemented to limit over-abstraction of groundwater in the Tablas de Daimiel, Spain, only the full compensation of farmers' foregone benefits proved to be successful (in contrast, compulsory quotas were not). Agriculture is in any case heavily subsidized and it makes sense to redirect subsidies away from incentives that tend to increase pollution (e.g. by rewarding higher yields) to those that promote good environmental management. Delinking of subsidy payments under the CAP is undoubtedly the most important and dramatic example of this trend, with the major underlying objective of promoting environmentally sustainable agriculture throughout the union (Berbel *et al.*, Chapter 13, this volume).

In conclusion, as in the case of opportunity cost pricing, there are severe practical difficulties of estimation, implementation and enforcement on the one hand, and of persuading farmers that they should pay for environmental externalities that – in their view – have only a tenuous connection with their activities on the other (World Bank, 2003a,b). Direct treatment measures can perhaps be 'internalized' but, with little agreement on how broader externalities can be valued, there is little prospect that farmers will be persuaded to pay for what they do not regard as their responsibility, and little prospect that politicians will impose such burdens under conditions of rising income inequalities and farmer unrest.

SYNTHESIS: CONTEXTUALIZING THE DEBATE AND SUGGESTING ANSWERS

An Emerging Storyline

This chapter has reviewed the different objectives of water pricing policies in agriculture. The overall picture that emerges is that of a gap between stated objectives and expected benefits on the one hand, and the actual and foreseeable impact of these policies on the other. Too often, stated objectives are based on analogy with the water supply and energy sectors. However, such an extrapolation can be very misleading given the particular characteristics of the irrigation sector.

An assumed correlation between low charges and low efficiency in surface irrigation has fuelled the chief narrative on water pricing. From this alleged causal link, it is inferred that raising prices would generate more careful practices and efficiency gains. Although generally valid for water supply and energy, this cannot be systematically assumed in irrigation. Reasons, in part, reflect the hydrological context and the characteristics of irrigation design and performance. In practice, most schemes and farmers are 'water takers', using whatever water is supplied to them, with the causes of uneven and unpredictable supply typically lying upstream of the scheme. Even when scheme supplies can be assured, it is deficiencies in scheme management that result in uncertainties and inequities at the farm gate rather than any price (dis)incentive. Farmers' responsiveness to price requires that charges are volumetric. Farmers have control over the quantity of water they take and the price is sufficiently high to correspond to the elastic portion of the demand curve. This combination of circumstances is, unfortunately, exceedingly rare.

Empirical evidence suggests that under conditions of scarcity: (i) farmers use water more efficiently, in particular, through conjunctive use; (ii) basin-level efficiency rises considerably; and (iii) surface water use is almost invariably regulated – in a more or less controlled manner – by rationing and quotas. The prevalence of quotas can be explained by their effectiveness in balancing supply and demand in response to variable supplies, while incurring far less loss in income than with price-based regulation; their relative transparency and equity; and the low infrastructural and transactions costs involved in their establishment. In a few modern systems, users have some latitude to use water above (or below) their quotas and in these cases water charges can be effective in influencing use at the margin. Markets at local level can also help balance supply and demand. Wider markets in quotas (water rights) can also promote high-value use, but have demanding technical and institutional preconditions and are seldom feasible in practice.

A more profound change than any of these has, however, been the spread of tube wells. By allowing farmer control, tube wells offset the risks, inadequacies and uncertainties not only of rainfall, but also of surface supply. Not only does this approximate to irrigation on demand – the holy grail of advocates of modernization and water pricing – but it also detracts from the need to deliver water on demand in surface systems since groundwater irrigation can (and in practice does) support a large part of the crop diversification and high-value farming that can be realistically envisaged. Ironically, and in contrast to surface supplies, it is the transaction costs of enforcing quotas that is prohibitive in the case of groundwater, and it is the long-term degradation of the resource that represents the major challenge in groundwater management.

What then is the role of irrigation water charges in surface irrigation? Figure 2.9 repeats the objectives suggested in Fig. 2.1, together with a summary of the constraints on achieving these objectives that have emerged in this chapter. They are briefly discussed below.

Economic theory suggests that, if the necessary preconditions are met, marginal cost pricing provides the signals to the farmer that optimizes his use of water. In contrast to the water supply and energy sectors, this chapter has suggested that marginal costs in irrigation should generally exclude initial capital costs. If so, direct marginal costs as a minimum comprise recurrent O&M, replacement and modernization costs. In principle, they should also reflect opportunity values in other uses and incorporate externality costs. The estimation and implementation of these measures is, however, fraught with difficulties. Moreover, marginal cost pricing is dependent on volumetric control, and in practice, pricing of water falls well short of full on-demand pricing.

Recovery of O&M costs is the most compelling reason for levying irrigation charges, notably if public funds are insufficient to operate and sustain the infrastructure. Cost recovery has understandably been the central objective of project design and national policies, and has become more pressing as irrigated areas have expanded and fiscal constraints have developed in many countries. Recovering just O&M costs has, however, proven much harder than expected and in the great majority of cases farmers are charged no more than a share of these costs. Moreover, defaulting is pervasive, especially in systems where supply is unpredictable and uneven and where staff has no incentives to enforce recovery. In a few cases, a share of capital cost is also recovered in addition to O&M, and/or farmers pay a management or a resource fee, or

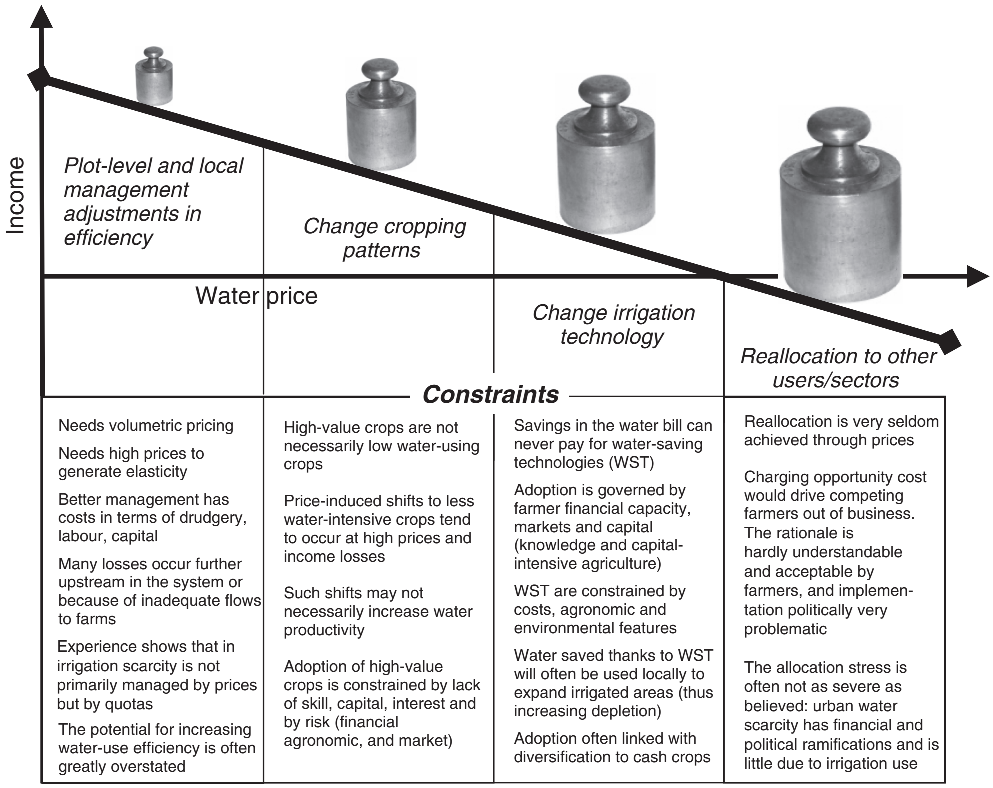

Fig. 2.9. Summary of constraints to using prices as an economic tool.

an environmental tax, but these seldom total more than about 10–25% of O&M costs. Charging for capital costs in new projects has the potential to ensure cost-effectiveness and users' interest and to crowd out politically motivated projects, but this is as yet seldom applied.

A wide array of benefits beyond sustaining the infrastructure is often anticipated for water charges, even when not warranted by the level or structure of the charge. This may reflect an improper understanding of charging mechanisms or be a means to justify the proposed policies. Chief among these are the view that raising prices will contribute to water conservation though, as discussed above, this is seldom valid. Charges may, however, have potential for eliciting longer-term shifts in crops and technology. Farm models often suggest that price-induced shifts and attendant water savings are possible but, as in the case of reducing water use, crop and technology choices are usually determined by other factors. Poor farmers irrigate low-value crops for many reasons (risk, capital, skill, markets, water supply, etc.) and, in particular, the risks to them of shifting to higher-value crops are considerable. Moreover, high-value cropping is inherently limited by market conditions and surface irrigators must compete with those having access to tube wells. If alternative crops or possible gains in efficiency are limited, farmers with extensive agriculture and low revenues will often revert to rain-fed farming, rent or sell out their farm, or just keep land fallowed, unless subsidies help them invest and intensify their practices. In practice, subsidies are often made available for such farmers.

High-value cropping often goes together with modern technologies, taking advantage of a host of positive factors beyond water savings, including higher yields, better product quality, fertigation, reduced labour, etc. Water costs are seldom the only or even the primary motivation for such shifts. In addition, water-saving technologies reduce return flows, but impact little on the fraction depleted by evaporation and transpiration; and in some cases, the water saved is used to expand the cultivated area, thus increasing depletion. In the latter case, promoting micro-irrigation can be counterproductive since the fraction consumed by crops increases at the expense of aquifer recharge, return flows and/or reallocation to other uses.

Low charges are also commonly taken to indicate a misallocation of resources that can be rectified by charging an opportunity cost. In practice, not only has opportunity cost pricing seldom, if ever, been attempted, but the very existence of an 'allocation gap' can be disputed. Priority is invariably given to M&I during a drought; over the longer term, most countries transfer water out of agriculture by stealth or administrative action; and there is little to indicate that urban and economic growth are eventually seriously constrained by water that is locked up in irrigation uses (except for some situations in the USA). Urban water and sanitation deficiencies are overwhelmingly due to political priorities and financial constraints rather than to lack of water. Moreover, opportunity cost is location-specific and, once effective demand in competing M&I uses is satisfied, opportunity cost falls off drastically. Opportunity cost pricing would drive those few farmers facing urban competition out of business, while most others would continue to obtain water at a much lower price. Markets are an attractive alternative, but the technical and institutional preconditions are daunting. Perhaps the most promising approach is negotiation on a case-by-case basis since, though government regulation is still required, compensation can be assured to those deprived in an open and transparent manner and in ways adapted to the particular conditions. Planning compensation mechanisms for temporary transfers *in anticipation* of drought will help avoid conflicts and turmoil when these occur.

Similar practical objections face the estimation and implementation of environmental pricing. Any charge that limits water use is likely to have some positive environmental impact but, given the constraints discussed above, imposing additional environmental charges on water use may not be feasible. It is therefore, perhaps, no surprise that while both the user-pays and the polluter-pays principles claim to internalize externalities by negative incentives at the source, in practice these externalities tend to be internalized at the system, basin or national level, through cross-subsidization from other users or the general taxpayers. Users get paid to control water losses or pollution, or even for the foregone revenue of not creating the externality, rather than being charged for the externality.

In conclusion, given the struggle to recover O&M and other recurrent costs in large-scale public irrigation, it is unlikely that water charges at levels much above O&M costs will ever become feasible. Participatory management, co-management, and autonomy can strengthen incentives for meeting the financial costs of supply, but irrigation charges are unlikely to have major impact on cropping patterns, technology or allocation between sectors; objections to opportunity and externality cost pricing will remain and, where farmers are given a say in the determination of charges, these are unlikely to be set much over O&M costs. In sum, whether management remains under state agencies or is shifted to farmer organizations, O&M will remain the reference 'peg'. Pricing will be sometimes effective in groundwater use and as a mechanism to regulate use beyond the quota, wherever individual volumetric pricing is possible. Bulk allocation with innovative incentives may also, in the future, help achieve efficiency gains, as experimentation in China suggests. In other words, the consensus of the mid-1980s (see Molle and Berkoff, Chapter 1, this volume) still largely holds and much of the discussion on pricing instruments in public surface irrigation, and the hopes vested in them over the last two decades have been an unhelpful distraction. Physical sustainability and proper management remain compelling objectives and finding ways to strengthen financial autonomy and the reliability of supply remains paramount.

Cost Sharing with Power Sharing

Analysts in the 1980s appreciated that irrigation pricing policies had limited potential for promoting conservation and reallocation. Rather, they emphasized that farmer payments should be part of a wider realignment of roles and responsibilities in irrigation management. Irrigation charges could be the 'glue' of contractual arrangements between higher- and lower-level entities, down to the WUA. Autonomy at each level would create 'downward accountability', with payment made from the lower to the higher level in return for a negotiated service (defined as a certain pattern of supply). Each level would maintain and operate the infrastructure under its jurisdiction while contributing its share of system O&M costs. Under such conditions, user charges could help: (i) enhance availability of funds for O&M; (ii) strengthen accountability of managers to water users; (iii) increase involvement of water users in O&M; and (iv) improve the quality of investment decisions (Small, 1990).

This model has been constantly rediscovered and is deeply interwoven with strands of participatory management and turnover (Molle and Berkoff, Chapter 1, this volume). The nature and scale of what is transferred have varied widely. In some cases (Thailand, Sri Lanka, Pakistan and India) participation was based on tertiary canal user groups that were to federate. In practice, however, most were given too little power and fee collection has often failed (Merrey, 1996). Limitations in hydraulic infrastructure (Lankford and Gowing, 1997; Facon, 2002) have also been a constraint that often revealed the mistaken conception – perhaps inherited from domestic water supply – that it is possible to define a service in irrigation as 'simply' as in the domestic sector. In more successful cases (Mexico, Turkey and Argentina) O&M of the main system are retained by the public agency but WUAs are established at block and tertiary levels. In yet other cases, often smaller schemes with fewer richer farmers, the scheme has been entrusted wholly to farmers, with the state retaining a supervisory role (e.g. in Peru: Vos, 2002; Colombia: Vermillion and Garcés-Restrepo, 1998; Japan: Sarker and Itoh, 2001; and Catalonia: Fernandez-Urrutia, 1998).

The responsibilities transferred have also varied. WUAs are generally responsible for O&M within their area of jurisdiction, but some are only responsible for water management at higher levels. Their role in planning may be symbolic (allocations decided by the agency based on water availability), more proactive (with joint decisions on allocations to different areas) or even entail total responsibility. Financial contributions also differ (Spencer and Subramanian, 1997). Allotments to WUAs can be decided by the agency alone or jointly with WUAs; enforcement and monitoring of service can be more or less strict and with varied recourse by users; WUAs may trade allocations (as in Mexico); and in some cases charges levied also fund part of the agency's costs, while in others the agencies are subsidized by the state. Variations are inevitable and desirable and it is difficult to generalize. Nevertheless, empirical evidence collected over the last 20 years or so suggests a number of observations on the basic pattern.

The model is by and large valid but has exceptions

There is a strong relationship between the power devolved to farmers and their financial contribution. Where farmers are confined to tertiary-level activities, success has often been poor. When given management responsibilities besides O&M, they have often been able to take more substantive decisions, e.g. hiring field staff and deciding how to spend funds on maintenance (Mali: Aw and Diemer, 2005; northern Peru:

Vos, 2002; Argentina, etc). Where they are also contributing to the costs of running the public agency, their powers also tend to increase (Peru, Colombia), though this is not always the case (Vietnam: Fontenelle *et al.*, Chapter 7, this volume; Philippines). A farmer's financial contribution to O&M is no doubt necessary if farmers are to be given significant managerial powers, but is neither necessary nor sufficient for effective overall management and maintenance. In some cases (e.g. Morocco, Tunisia and Iran) farmers cover most or all of O&M costs and receive a reasonable service without strict accountability mechanisms. In contrast, the NIA in the Philippines illustrates the dangers of overestimating the capacity of supposedly autonomous agencies to ward off political interference. Moreover, NIA has responded to inadequate funds not by augmenting revenues, but rather by reducing costs and servicing only parts of the system (Kikuchi *et al.*, 2001; Oorthuizen, 2003). In the case of Taiwan (Moore, 1989; Lam, 1996) effective management by officials and farmers is achieved though user charges have long lost their significance, since the state re-established O&M funding in the early 1990s. Accountability is not supported by bureaucratic rules, but is embedded in social relationships and social control.

Narrow functionalism

Small and Carruthers (1991) recognized 'linkages existing between structural and managerial aspects on the one hand, with financial approaches on the other' (Small, 1990) but retained a functionalist view of agency–farmers arrangements: that charging linked to accountability could ensure transparent and effective cross-compliance and end the 'degradation vicious circle'. They have been criticized for overlooking the wider social and political dimensions that affect the level and utilization of charges independently of performance (Oorthuizen and Kloezen, 1995). Water charges are elements of negotiation in power struggles between farmers and their associations, and between WUAs and the agency or state. While these negotiations are bounded by hard-nosed realities, such as farmer financial capacity and the actual cost of supplying water (Lee, 2000), they also reflect competing interests, differing perceptions, the political clout and bargaining power of the different parties, and the various levels of accountability and dependency between them. They are permeated by the distribution of power within and across these groups (see case studies for the Philippines: Oorthuizen, 2003; Peru: Vos, 2002; Vietnam: Fontenelle *et al.*, Chapter 7, this volume; Taiwan, South-Korea, Japan: Sarker and Itoh, 2001; Tanaka and Sato, 2003). In other words, while 'money talks' and creates some dependency, accountability was shaped predominantly by inter-group and interpersonal relationships expressed in such factors as friendship, kinship, gifts, business partnerships, bribes, threats of violence, patronage, debts, asymmetries of power and information, and political allegiance. This warns us against simplified views of human organization and may help anticipate dysfunctions.

Second-generation problems

Encouraging financial and managerial autonomy of irrigation blocks or schemes coincides with the retreat of public agencies to higher levels of management. Autonomy has, in general, been successful in divesting the state of financial burdens but, according to many observers, has been largely neutral in terms of irrigation efficiency, water reliability and water productivity (Meinzen-Dick *et al.*, 1994; Vermillion, 1997). This in part reflects unrealistic expectations given that irrigation has always been more efficient than is commonly supposed and that farmers and managers have in any case adjusted to prevailing conditions. But it also reflects 'second-generation problems' that have gradually surfaced and have adversely affected performance including: the failure to adjust charges leading to deferred maintenance; the lack of data collection and analysis; imprecise rules governing asset ownership and management; and an unclear definition of water rights (Svendsen *et al.*, 1997; Vermillion and Garcés-Restrepo, 1998; Vermillion and Sagardoy, 1999). Among these, the most important problem has probably been the first: a short-term unwillingness to adjust fees upwards, to the detriment of long-term sustainability.

Opening up the model

The focus on financial autonomy has sometimes been superseded by more general participatory policies that emphasize reducing agency costs, or social engineering objectives. Nevertheless, there has also been renewed interest in the potential role of private operators and public–private partnerships (Frederiksen and Vissia, 1998) and in reviewing the whole spectrum of 'water service entities' from private to self-governing bodies (Lee, 2000; ICID, 2004; Frederiksen, 2005). Préfol *et al.* (2006) have pointed to the need for 'professional third parties' between farmers and government, irrespective of whether these are public or private. The crucial questions are accountability and incentive structures (Merrey, 1996). Promotion of volumetric management and bulk allocation is no doubt essential, but cannot ensure that incentives reach the individual farmer. Greater attention thus needs to be given to strengthening incentives at the tertiary and block levels. Interesting examples include the Philippines, where commissions are paid to WUAs that are successful in recovering charges (Ofrecio, 2005), and China where managers and subcontractors have both been given performance incentives (Lohmar *et al.*, Chapter 12, this volume; Li, 2006).

An alternative to the fiscal autonomy model patterned on utilities (O'Mara, 1990) takes up the idea of water delivery as 'co-production' (Lam, 1996; Ostrom, 1996). Under a 'co-production' approach, farmers and others participate in the production of public goods, in contrast to a 'service' approach under which they are merely passive 'clients'. It is argued that involving users at higher levels strengthens accountability and ensures that participants are aware of management constraints, existing inequities and actual available resources, the aim being to shift their role from that of 'selfish complainers' to co-managers of the whole system. According to this, the state must still inevitably retain supervisory powers, especially over financial management and maintenance standards, and in this regard it is lack of effective government capacity rather than lack of farmer and 'client' awareness that remains the major obstacle to creating self-sufficient entities (Frederiksen, 2005).

Perspectives for the Future

This review suggests that water charges can only achieve the objectives assigned to pricing as an economic tool (Fig. 2.1) in very special circumstances. But there is a continuum from projects with excess water and poor management at one extreme to those under volumetric management and – at the limit – irrigation on demand, at the other. Scarcity will continue to be dealt with by rationing in the large majority of cases, but price incentives can sometimes promote conservation and in a few cases regulate water use at the margin. The way forward is thus to expand the area served by volumetric management so as to facilitate extension of quota-cum-price regulation (Fig. 2.10), recognizing that this will be a slow process, given the structural and institutional changes needed, and that it may not always be appropriate or cost-effective to do so.

Such changes cannot be driven primarily by modernization investment or by social engineering that is inconsistent with the broader context. Effective financial mechanisms are predicated on the emergence of autonomous entities that vary with context but which entail genuine user empowerment. It should be recognized, however, that irrigation efficiency and water productivity are more about changes in irrigation management than changes in farmer behaviour;

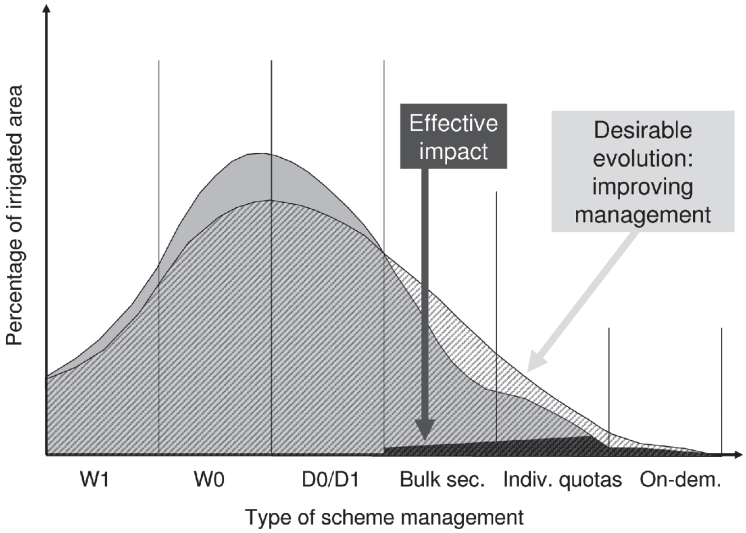

Fig. 2.10. Management types and desirable shifts.

more about designing cross-compliance arrangements and financial autonomy than simply establishing WUAs; (iii) and more about defining positive incentives to managers than introducing negative incentives to end-users.

Policies based on negative incentives alone are unlikely to have great success. The user-pays and polluter-pays principles thus need to be complemented by positive incentives. It may be more efficient (as well as more equitable) to buy out wells than to decree extraction quotas; to pay upstream farmers for not polluting water or deforesting watersheds than to tax these activities; and to negotiate compensation arrangements for water transfers than to expropriate them. The limited capacity of the state, and the political sensitivity of actions to modify behaviour that result in significant loss of income are major reasons why water and pollution charges have, in practice, been so difficult to introduce and enforce. Policy packages should ideally combine 'positive' and 'negative' instruments in ways that are adapted to circumstance (Bazza and Ahmad, 2002; Chohin-Kuper *et al.*, 2002; World Bank, 2005a). Since many factors other than water price so often determine water use, water policies must also be designed with due consideration to policies in other sectors.

Since individual metering is so problematic in surface irrigation, priority must be given to bulk allocation, all the more because it is consistent with strengthening co-management institutions and arrangements. Since financial incentives seldom impact directly on individual users, emphasis should normally be placed on management incentives (whether to private or community operators), while ensuring financial transparency. This is consistent with the fact that efficient management of supply is easier at block level than at individual farm level. There may be potential for trading in bulk allocations within the system, provided this is ultimately decided by stakeholders and can be effectively regulated, but intersector trading is likely to be feasible in only a few exceptional circumstances.

It must be recognized that much, if not most, surface irrigation, especially in countries with large irrigation sectors, will continue to be devoted to cereals and other relatively low-value crops. No doubt an increasing number of farmers will intensify and diversify output, often based on tube wells, but this is limited by market constraints and most farmers in surface irrigation are likely to remain relatively poor, at least as long as prices remain at current levels and until such time as economic development draws population off the land sufficiently to allow significant farm consolidation. This suggests caution in implementing expensive modernization and similar programmes that may not be justified by the production benefits. It also suggests the necessity of taking account of the deep social and political concerns raised by poor farmers. As stressed by Garrido (2002): '[N]o pricing policy will ever make progress if irrigators' benefits are severely compromised as a result of its full implementation. In the short and medium term, irrigation farms' economic survival is essential.' Economic policies pursuing efficiency will thus inevitably have to compromise with equity and social concerns and take into consideration the diversity of farming systems and regions.

Overemphasis on 'getting the prices right' (Svendsen and Rosegrant, 1994) has distracted attention from the nature of most of the irrigation in developing countries. Very few schemes can distribute water in a way approaching the on-demand supply model that typifies urban tap water. Farmers cannot be blamed for losses occurring upstream of their farm; nor can they be blamed for much of the waste arising out of a pattern of supply that is largely independent of their will. The importance of the old unglamorous issue of *managing supply* will thus continue to override that of *managing demand*. No doubt this will gradually change as irrigation moves along the continuum suggested in Fig. 2.10. But even then, developed countries' experience suggests that most efficiency gains are due to the numerous other factors involved in the shift from pragmatic to volumetric management; and that the task left to pricing even in the long term may well be far more modest than often assumed.

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