PII: S0165-0203(97)00010-X

Natural Resources Forum, Vol. 21, No. 3, pp. 191-200, 1997

0165-0203/97 $17.00 + 0.00

Reflections on the meaning of sustainable development in the water sector

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Economic interpretations of sustainable development have concentrated on the need to maintain the aggregate capital stock intact, so as to ensure a constant stream of welfare through time. A sectoral approach to sustainability, in terms of this model, will be justified if substitution possibilities between that sector and the rest of the economy are limited. This assumption is examined in relation to the water sector, and it is found that further assumptions about substitution possibilities need to be made within the sector itself. Suggestions are offered as to how such substitutions might be made to advantage, which is contrasted to recent water management practices in England and Wales. It is argued that assumptions about substitution possibilities are in general overly restrictive, with the result that sustainability objectives fall short of optimum levels that could be achieved. © 1997 United Nations. Published by Elsevier Science Ltd

Since the publication of the Brundtland report (World Commission on Environment and Development, 1987), "sustainable development" has been declared the basis for policy formulation in many government organizations and agencies in the U.K. and elsewhere. For instance, the 1990 White Paper *Our Common Inheritance* identifies sustainable development as a central tenet of U.K. government environmental policy. Further, by becoming signatories to Agenda 21 at the UNCED Earth Summit in Rio de Janeiro in 1992, many governments supported the establishment of the Commission for Sustainable Development, and committed themselves to preparing national strategies for the implementation of Agenda 21 agreements. Finally, the National Rivers Authority (NRA), the body responsible for the management of water resources in England and Wales until last year, had a stated objective of applying the principles of sustainable development to the management of water resources and water quality (NRA, 1991a).

Sustainable development has been broadly interpreted at the level of the macroeconomy to refer to some suitably defined measure of national well-being—often an adjusted form of standard national income—being at least constant over time, and preferably rising (hence, "development"). More completely, sustainable development also commonly includes some reference to the *current* distribution of income and resources becoming no more unequal as development occurs. The origin for this interpretation, at least within the economics literature, was John Rawls's (Rawls, 1971) *A Theory of Justice*, which advocated the so-called "max-min" approach to social choice, which in turn strictly implies equal well-being through time and across social groups (Solow, 1974). Hence, sustainable development addresses issues of both *inter-* and *intra*-generational social equity. As such, then, it is not an objective criterion by which to judge policies and policy outcomes, but implies a particular value judgement on the part of the decision-maker (as, indeed, do all criteria for choice). The question then might be whether the decision-maker considers it appropriate that s/he should be arbiter of such apparently socially-important questions.

Unfortunately, research into the sustainability question is not widely recognized or understood in policy-making circles. Inefficient and ineffective policy-making could result, implying two things: that the outcome of a policy could be improved upon without any extra cost (e.g. water quality could be higher for the same commitment of economic resources); and/or that the quantities of other valuable economic goods and services could be increased without the need for a lesser policy outcome (e.g. taxes could be lowered whilst still achieving the same level of environmental protection).

In the economics literature, the most commonly investigated sustainability model centres on the need to maintain the *aggregate* capital stock intact. In this

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paper, we examine the conditions under which *sectoral* policies for sustainable development are justified. These conditions enable us to arrive at some simple rules for defining the meaning of sustainable development in the water sector. However, they also highlight some difficulties in these rules. We illustrate these difficulties with particular reference to water management in England and Wales. We also argue that these difficulties could lead to a "double failure": not only might the sustainability objective arrived at be suboptimal, but the policies formulated to achieve that objective might be inefficient also.

The economics of sustainable development

We can apply the term "capital" to any durable asset which provides valuable services. Conventionally defined, an economy's capital stock includes buildings, plant and machinery and so on. But clearly, the environment as a whole also provides us with an overwhelming range of goods and services, ranging from the simple recreational pleasures of woods and parkland, to the vital life-preserving functions of the ozone layer and the carbon cycle. These environmental resources can potentially provide their services *in perpetuity*, whereas others are effectively limited and finite (e.g. oil). But all are economically important, since the services they provide are valuable and contribute positively to human welfare and well-being. The interpretation of environmental resources as "capital" follows naturally. Then the well-being of any generation depends on the value of the services flowing from the *total* capital stock existing in the economy at that time, which in turn depends on the capital stock's size and mix. "Natural"—or environmental—capital plays a greater role in some economies than others. The World Bank (1997) recently estimated that only 1% of Germany's total wealth was accounted for by natural capital, whereas that figure rises to 18% for Cote d'Ivoire, and 54% for Niger.

If well-being derives from the consumption, in the broadest sense, of goods and services provided by an economy's capital stock, and sustainable development requires well-being to be at least constant if not increasing over time, then sustainability also requires the capital stock to be maintained or increased over time. Maintaining the overall capital stock intact would seem to ensure that future generations will be able to enjoy the same well-being—generating possibilities as those of the current generation (Solow, 1986). Capital stocks decline because new investment in them is insufficient to compensate for any depreciation. Therefore, maintenance of the capital stock requires no net depreciation over time; increases in the capital stock require positive net investment over time—either in the quantity of capital, or in technological innovation to increase its efficiency. Given that investment is simply the difference between output (net of depreciation) and consumption, then increased investment implies reducing consumption. This explains the focus on aggregate savings of Atkinson *et al.* (1997) and the World Bank (1997) (see below).

Sustainability, substitution and the capital mix

A difficulty arises when we acknowledge that the economy's overall "capital stock" is not homogenous but in fact comprises many different "substocks". Broadly speaking, these might include the air we breathe, fresh water, forestry, biodiversity, plant and machinery, cultural assets and buildings, mineral resources, fossil fuels, fisheries and so on. It is unrealistic to try to guarantee that each and every one of these substocks should be maintained over time. In fact, in the case of (effectively) finite resources such as oil, or fossil groundwater, this would imply some capital remaining essentially unused through time, which would seem to be clearly undesirable. Indeed, to a large extent, the value of resources such as these stems directly from their use—they have no real *intrinsic* value. A sustainability constraint which prevented the use of these resources, and thereby rendered them economically worthless, would be nonsensical.

However, this difficulty can be overcome if we recognize that not all types of capital are crucial to human well-being; i.e. some types of capital have substitutes. Then, reductions in some types of capital can be allowed so long as we ensure other types of capital increase in quantity or quality (Solow, 1974; Hartwick, 1977, 1978a, b). This extends to the environment also. Environmental damage caused by coal-mining, for instance, could be compensated for by increases in the industrial base or improvements in natural amenity elsewhere. This compensation might come about via a microeconomic policy of "compensating projects" specifically designed to offset particular instances of capital decumulation (Barbier *et al.*, 1990). A high-level alternative would be a "disembodied" macroeconomic policy which ensured that net investment in the economy was positive (Atkinson *et al.*, 1997). Consistently negative net investment at the macroeconomic level is indicative of non-sustainability and a declining capital stock (Pezzey, 1994). The World Bank (1997) has recently presented an attempt at applying a framework to measure such net investment, or, to use the World Bank phraseology, "genuine savings"—the difference between investment—in man-made and human capital, and environmental quality—and capital depreciation, which includes some environmental damage "monetized" through the estimation and application of "shadow prices" for environmental resources. Although still very limited in its coverage at this stage (mineral and fuel resources, timber, carbon dioxide damage, and education are the "non-market" investments considered so far), the exercise does serve to demonstrate the relative importance of environmental damage in comparison with more conventional concerns, especially in certain regions of the world (for instance, Sub-Saharan Africa). And if a country exhibits negative genuine savings under the current limited framework, then it will certainly continue to do so under a more extended one.

But the usefulness of "genuine savings" type measures and the feasibility of both compensating projects and macroeconomic savings policies as means to achieving sustainability, rest on the extent to which capital stocks actually *can* substitute for each other. The idea that new sources of fuel, perhaps wave power, or more fuel-efficient machines can compensate for reductions in the stock of oil might not be controversial. Perhaps less obviously (but no less reasonably in principle), the reduction in amenity and recreation possibilities stemming from the loss of a piece of woodland might be offset by the construction of cultural monuments or other assets.

However, some contend that the environment *as a whole* underpins the workings of the entire economy in a fundamental way, so that, in the limit, its destruction cannot be compensated for (Pearce *et al*.. 1989). Indeed, the functions of the environment are so complex, and our knowledge of them so hazy, that *any* damage is accompanied by a significant degree of risk and uncertainty (Pearce *et al*., 1994). The result of these considerations is the call for *strong* sustainability. Strong sustainability also requires the stock of *environmental* capital to be at least constant over time. Thus, a development path which involves a *systematic* decumulation of the stock of natural capital is not considered sustainable according to this criterion, even if the man-made capital stock is increasing at the same time. Environment-manufacture substitutions can still be made, but only under the condition that no net environmental damage occurs in the long run.

Other commentators, however, go even further than this, arguing that certain individual environmental resources are *critical* to both the economy and the environment. These resources do not have any substitutes, so that no depletion of or damage to these stocks can be compensated for by increases in stocks of other types of capital, whether natural or man-made (Mäler, 1986). Which resources match this description is open to debate, but Pearce *et al*. (1994) have suggested that ecosystem services such as the assimilative capacity for industrial wastes, the supply of biological diversity, the role in modulating climate and maintaining clean air and water, the maintenance of fertile soil, and so on, are seen by ecologists as representing the natural world's "life-support systems" for which few real substitutes exist. An economy cannot possibly be sustainable if these critical resources are being depleted or damaged, no matter what the level of investment elsewhere. This would appear to imply a need for sectoral sustainability policies, or sectoral monitoring at the very least.

In fact, this is the way many environmental policies for sustainability are often formulated in practice. The second UN ECE Sulphur Protocol has the ultimate aim of achieving soil and water *critical loads* for acid deposition in Europe in the long run, i.e. levels at which no effect (whether harmful or not) on soil and water chemistry can be discerned. The NRA has the statutory objective of ensuring sustainable use of the water environment, although no specific interpretation of this objective has been provided in official documents. Elsewhere, policies for sustainable agriculture, sustainable housing, sustainable communities and even sustainable citizenship are advocated (e.g. Lang and Hines, 1993). But, at least in terms of the economic model of sustainability, many such policies seem to imply particularly arbitrary definitions of what constitutes "capital". They also do not appear to be based on any explicit (or, often, realistic) assessment of the options available for substituting environmental damage with capital improvements elsewhere.

Sustainable development and the sustainable use of critical resources

It could be argued that some activities which use water as an input are inconsistent with sustainable development as we have defined it above. Nuclear power might be such an activity because of the potential severity, long time-horizons and general uncertainty surrounding the costs associated with the generation and disposal of radioactive waste. However, this says nothing about whether nuclear power is consistent with objectives of sustainable water management, which is concerned only with environmental damage in the water sector, not elsewhere. "Critical sustainability" should only be viewed as a necessary, but insufficient, condition for aggregate sustainability.

In other words, for the current case, if fresh water can be deemed a form of critical capital, then sustainable water management is necessary for both sustainability in the water sector and sustainability at the level of the aggregate economy. It cannot on its own, however, guarantee aggregate sustainability. On the other hand, if fresh water does not match the definition of critical capital, then sustainable water management is necessary neither for water sustainability (which becomes largely meaningless) nor aggregate sustainability. Indeed, following a policy of sustainable water management which was not justified by available substitution options could generate significant income or output losses. The question then becomes whether fresh water should correctly be treated as a form of critical capital.

Sustainable development and economic efficiency

The framework on which the constant-capital model is based assumes that standard economic efficiency requirements are met. Thus, economic decisions and policies still need to take into account the *full social costs* of their outcomes and impacts. Indeed, sustainability would seem to imply *higher* shadow resource prices than are implied by even the standard economic model (Pezzey, 1994). Decisions to license water abstraction still need to recognize the potentially high costs of building new reservoirs. Policies to control water pollution still need to account for the fact that effluent discharges can reduce the value of fisheries, and lower amenity values in river corridors.

Hence *sustainable development should not be seen as a replacement for standard economic analysis, but rather as an important supplement and qualification to it*. This raises the possibility that although policy outcomes might be sustainable in simple terms, their achievement might actually be *economically inefficient*. If policies for sustainability are formulated without a full input of economic analysis, the resulting level of well-being, even if sustainable, will not be maximized. An inappropriate assessment of the substitution possibilities associated with capital resources might be a particular source of such inefficiency.

Sustainable water management

Fresh water as critical capital

The preceding discussion suggests that, according to the constant capital model, a policy of sustainable water use is only justified if fresh water constitutes an example of critical capital, *or* if there are other reasons which make singling out fresh water for individual attention an efficient part of a policy for aggregate sustainability.

We might consider fresh water an example of "critical capital" on the grounds that its availability is self-evidently a pre-requisite for human life. Fresh water underpins the operation of the planet's very ecosystem. It is quite clearly an ethical imperative that current use of fresh water should not undermine the global ecosystem, as this is likely to have catastrophic consequences for future generations.

But it is also clear that current levels of water use in the U.K. and elsewhere far exceed those levels which might be necessary for the simple sustenance of human life. However, we might still justify singling out fresh water as a candidate for individual policy attention on the following grounds. Consistent with concepts of "relative poverty" (e.g. Townsend, 1979, 1985), water use which might appear luxurious at low income levels can become necessary at higher levels of income because of the changing sociological requirements for an individual to be able to play a full role in society. Expectations about how an individual should, and should be able to, behave in society change as that society develops and becomes richer. Hence, uses of water which are not essential to the maintenance of life may be crucial to an individual's being a free and accepted member of society. The availability of water for personal hygiene, or perhaps for washing clothes, might be an example. Those uses might then become "critical" in a social sense to individual well-being. Moreover, as a society develops, and incomes rise, the technological relationships which define the way individuals are able to satisfy basic human needs can change, and in particular, the availability of simple and affordable technologies can be restricted. For example, individuals on low incomes might find it increasingly difficult and expensive to wash their clothes, as household penetration of automatic washing machines increases as average income does, and the provision of municipal launderettes declines.

In fact, whether we are able to specify water as critical capital or not, this does not lessen the value of examining the principle of sustainable water management, for this examination itself might go some way to answering the definition question.

Defining sustainable water use

If we accept the "critical capital" definition of fresh water, we need then to determine what this might mean for practical management purposes. We can define quantity-sustainable water use by reference to the hydrological cycle and the nature of fresh water as a renewable resource. According to the criteria set out above, sustainability requires that current water abstractions should not impose costs upon future generations. The quantity of water that is available for use in any particular period is equal to the difference between total precipitation and the amount lost through evapotranspiration ("effective runoff"), plus any water held in surface or underground storage. Our sustainability rule becomes: net water demand (accounting for water return and reuse) should be met out of effective runoff only. Such water use is clearly sustainable because it does not rely on any finite stocks for support.

There is a clear need to account for dimensions of the water stock other than the purely quantitative one. One of the most important of these is water quality, for, even if levels of physical water use were to remain within our sustainability limits, a general decline in quality would lead to a restriction on the range of uses for which water was suitable. Given the non-substitutability of water—and it would appear reasonable to contend that it is the demand for *high quality* water which is most inelastic—then this restriction can be viewed as equivalent to a reduction in the size (or, at least, the value) of the capital stock, which, we have argued, is inconsistent with sustainability. In fact, this elementary reasoning allows us to arrive at the simple sustainability rule: water quality should not decline over time. If we assume that individuals' preferences and perceptions are stable over time, then this will ensure that the water system remains suitable for broadly the same set of economic uses in the future as it is currently.

The unit of analysis

Our definition of sustainable water use immediately begs a number of questions about the appropriate unit of analysis. Firstly, we need to ask what is the appropriate geographical unit upon which we wish to impose our sustainability constraint. Clearly we could require water abstraction to be sustainable at the level of the individual tributary, or even the 100 m river stretch. Then net water abstraction rates at all points in the river system should not exceed flow rates at those same points. Alternatively, we might permit "locally" unsustainable water abstraction to be compensated for by increases in water flow elsewhere, for instance, via effluent discharge. (This happens already anyway to some extent, as abstracted water is never returned to the river system at exactly the same location.) Similarly, decreases in water quality in one area might be compensated for by quality improvements in another area. Alternatively, we might require the quality of each kilometre length of waterway to be constant over time. Hence, the question is one of to what extent water is *spatially substitutable*.

Secondly, we need to determine the appropriate length of time over which the sustainability constraint should be binding. Again, we might require water abstraction to be within the limit of effective runoff at all points in time. In the limit, this might imply abstraction only being permitted when it was actually raining, and that water use was instantaneous, so that the abstracted water could be returned immediately to the river system, rather than after some delay! More realistically, the constraint is likely to be set seasonally, or yearly. This would imply less period-by-period variation but greater "temporary" non-sustainability. On the quality side, the assimilative capacity (AC) of a receiving water can vary considerably with changing weather conditions so that quality could be expected to decline naturally and temporarily for a given level of effluent discharge. If water quality is deemed to be poorly substitutable over time, then this will mean that discharges will have to vary directly with what might be very rapid changes in AC. Otherwise, we might only be concerned with long-run trends in water quality, over years, decades or even longer. Analogous to the spatial question above, the appropriate unit will depend on the extent to which water abstraction is considered *intertemporally substitutable*.

We are not in a position to offer any definitive answers to these substitution questions. The discussion does suggest that our definition of water as critical capital requires consideration of the possibility not only that other types of capital might substitute for fresh water, but also that water might be able to substitute "for itself", at other times and in other places. This it quite clearly does: hence the practice of storing water in reservoirs for use at other times and in other places. Indeed, the use of reservoirs in itself suggests the acceptance of some short-run substitutability, and the non-sustainability that can be implied: water users must for some period be reliant on finite stocks located in other geographical areas.

However, it is clear from the literature (e.g. Pearce *et al.*, 1994) that sustainability needs to be seen as a *long-run* constraint upon human economic activity. The relevant time horizon is obviously not infinity, but it would certainly be a few generations, perhaps 100 years (*ibid.*). Then it is clear that it is the *long-run trends* in the water sector which are most important for sustainable water management as we have defined it. Thus, declines in water quality from year to year, or even over longer periods, should not be seen as necessarily unsustainable, so long as they do not become permanent and the long-run trend does not become negative. Similarly, current net abstractions can lead to temporary reductions in flow rates, but should not become dependent upon them. This characterization of intertemporal substitutability would seem to allow significant flexibility in the way water is managed over time in practice.

It is rather more difficult to extend this analysis to the question of spatial substitution. In the limit, it would seem to imply the quality of one body of water declining towards "zero" while another increases towards "infinity". In quantity terms, it might effectively imply the wholesale transfer of a body of water from one location to another. Intuitively we might have some difficulty with this interpretation, and the reason would appear to be linked with fresh water's status as a renewable resource. For instance, the ability permanently to transfer water from one place to another is limited by natural patterns of weather and physical geography, which provide a potentially irresistible bias back towards the original distribution. Similarly, a water body's natural assimilative capacity implies a natural tendency towards "positive" levels of water quality. As a result, it would seem that spatial substitution should perhaps be seen as a particular form of intertemporal substitution, which fits much better with prior intuition.

Although these considerations make it difficult to be categorical about the appropriate definition of critical water capital, it seems clear that current management experience suggests that a high level of abstraction might be broadly possible, permitting considerable management flexibility, especially in higher income countries. Moreover, it should be pointed out that some definition is likely to be determined *de facto* by the nature of water management in practice. For instance, the NRA (1991d) presents abstraction data annually, and limited by the boundaries of each NRA region. Annualized data can mask the considerable seasonal variation in rainfall, and sustainable prices based on them will tend to be *too low* in the summer months, when resources are relatively scarce, and *too high* in winter when most rainfall occurs. However, a sustainability view is purposefully long-term, and these short-run issues are probably better addressed on other, perhaps simple efficiency, terms. Further, a regional scale for analysis implies that issues such as the problems of low river flow experienced in many NRA regions in the past decade would be difficult to examine since they are essentially the result of a combination of local factors which regional data would tend to mask. This immediately suggests that a sustainability criterion of the type outlined above may be inappropriate for the consideration of a large class of potentially important environmental allocation decisions.

Water quality management in England and Wales

Policy towards water quality in England and Wales comprises essentially two instruments. Statutory water quality objectives (SWQOs) set the biochemical quality standards for each particular river stretch, which pollution control policy is intended to achieve. SWQOs are specified in terms of determinands such as dissolved oxygen (DO), biological oxygen demand (BOD), and ammonia concentration, and are usually set at a level consistent with the current uses to which the river is put. In some cases, however, they might specify a higher physical quality consistent with some "target" use(s). Uses which require higher physical water quality (such as salmonid fisheries, or potable abstraction) are assumed to correspond to those uses having higher economic value, although SWQOs are not tailored according to the likely demand for each potential use. Non-declining physical water quality is achieved by ensuring that SWQOs do not decline over time. Indeed, there is often a presumption in favour of a minimum target standard which all waters must meet in the longer term (Royal Commission on Environmental Pollution, 1992).

The substitution possibilities implied by this regime might be as follows. The condition that no single SWQO should be lowered over time implies that no spatial substitution is deemed feasible between quality standards of different controlled waters. Standards are commonly imposed upon river stretches of between 5 and 15 km in length. Hence, substitution is permitted within these limits but not between, i.e. no improvements in water quality, however large, can compensate for any decline which is more than 15 km away. Finally, Section 105 of the Water Act 1989 requires that quality standards be met "on and at all times" after the date set (NRA, 1991b), implying that, save for cases resulting from sampling error or unforeseen circumstances such as adverse weather conditions (e.g. droughts), temporary reductions in water quality can never be compensated for by subsequent increases, however large they might be.

The extent to which this management regime is successful in achieving sustainable water quality management can be examined by considering the results of the U.K.'s national water surveys. National water quality in England and Wales is assessed by river kilometre every 5 years. Until 1980, classes were defined solely in terms of BOD, but since then, DO and ammonia concentration have been added. The levels of these determinands for each class are those needed to protect those uses of the watercourse regarded as more important, such as potable water supply and fisheries (NRA, 1991b). Table 1 summarizes the results of the last six surveys up to 1990. Although the figures are not strictly comparable due to changes in sampling and classification methods, the trend appears to be one of marginal improvement from 1958 up to 1980, with perhaps a slight fall-off since then. The earlier improvement is likely to be the result of a run-down in old-technology, heavy industries rather than any particular effort to increase quality. The more recent reduction, which appears to have been concentrated mainly in the south east and south west, can be explained by a number of factors, including changes in survey methodology, increased discharges from sewage works, and two hot summers in the late 1980s. It is also apparent from first inspection of the latest (1995) quality survey (using a further refined classification scheme) that these declines have not yet been reversed (NRA, 1996).

As a result, in terms of the criteria set out above, we might say that trends in water quality in England and Wales have been marginally sustainable over the last 40 years. Over the past decade or so, quality has declined slightly, although this might not be of any serious concern assuming the time horizon we have suggested is relevant to sustainability issues. Note, however, that the fact that the results of quality monitoring are reported in terms of river *length*, rather than volume, almost certainly overestimates the availability of clean, compared with polluted, water (Royal Commission on Environmental Pollution, 1992). This is because the results are distorted by the many miles of small, fast-flowing, relatively unpolluted rivers which account for only a small proportion of the actual quantity of surface water. The impression of the severity or value of changes over time in the proportion of rivers in each quality class is therefore similarly distorted.

Note also the apparent potential for conflict between the pollution control and water quality monitoring regimes. As we have already seen, the former requires that no SWQO should be lowered over time, and that each SWQO should be met on and at all times. In turn, this implies a spatial substitution of between 5 and 15 km for any one river stretch, and no intertemporal substitution. The results of the river water quality surveys, on the other hand, are reported by river length every 5 years, which provides significant scope for variation in quality over time, especially given the limited management reaction to recent quality declines. Moreover, these results have tended to have been interpreted in an *overall* manner, with an increase in the proportion of total river length classified as, for example, "good" being interpreted as a general improvement (e.g. NRA, 1991b), which in turn suggests considerable scope for *spatial* substitution.

Our preceding discussion of substitution in the constant capital model of sustainable development might lead us to suggest that it is the quality survey characterization which is the more relevant for sustainability purposes. This characterization permits the type of flexibility which would appear to be a feature of that model. Indeed, it seems that only by considering the results of a number of surveys over an extended period of time can we garner any meaningful insight into how sustainable current water resource use is. The characterization provided by the SWQO regime, on the other hand, places significant restrictions on the way water resources are managed, restrictions which are almost certainly impossible to justify on sustainability grounds. This will severely limit the extent to which water resources can be used effectively in the economy.

However, whatever the current accuracy of assessments of the available substitution possibilities in the water sector in England and Wales, the resulting management objectives are unlikely to be achieved efficiently anyway. This is because of the nature of the policy instruments employed to control effluent discharges. SWQOs are achieved via discharge consents (i.e. licenses), with an attached charge determined by reference to a discharge charging scheme. The charging formula is given by:

$$\text{Charge} = A \times B \times C \times D$$

where A is the volume factor, B is the contents factor, C is the receiving water factor, and D is the financial factor. Each of the first three factors provide a relative charge weighting depending on the nature of the discharge and the consent conditions. For instance, a maximum daily volume of up to $5\,m^3$ is weighted 0.4, volume of between 100 and $1000\,m^3$ is rated 1.0, and volume over $150\,000\,m^3$/day is rated 14.0. Discharges which contain fungicides, herbicides or polychlorinated

Table 1 Water quality in England and Wales, 1958–1990

Former classifications									New classification							
1958–1980 Surveys									1980–1990 Surveys							
NON-TIDAL RIVERS AND CANALS									FRESHWATER RIVERS AND CANALS							
Class	1958		1970		1975		1980		Class	1980*		1985		1990		
	km	%	km	%	km	%	km	%		km	%	km	%	km	%	
Unpolluted	24 950	72	28 500	74	28 810	75	28 810	75	Good 1a	13 830	34	13 470	33	12 408	26	
Doubtful	5220	15	6270	17	6730	17	7110	18	Good 1b	14 220	25	13 990	34	14 536	34	
Poor	2270	7	1940	5	1770	5	2000	5	Fair 2	8670	21	9730	24	10 750	25	
Grossly Polluted	2250	6	1700	4	1270	3	810	2	Poor 3	3260	8	3560	9	4022	9	
									Bad 4	640	2	650	2	662	2	
									X	—	—	—	—	39	—	
									Unclass	—	—	—	—	17	—	
Total	34 690		38 400		38 590		38 740			40 630		41 390		42 434		
TIDAL RIVERS									ESTUARIES							
Unpolluted	1160	41	1380	48	1360	48	1410	50	Good A	1870	68	1860	68	1805	66	
Doubtful	940	32	680	23	780	27	950	34	Fair B	620	23	650	24	655	24	
Poor	400	14	490	17	420	15	220	8	Poor C	140	5	130	5	178	7	
Grossly Polluted	360	13	340	12	280	10	220	8	Bad D	110	4	90	3	84	3	
Total	2850		2880		2850		2800			2730		2730		2722		

Source: National Rivers Authority (NRA, 1991b).
*As revised in 1985.
Unclass, unclassified.

biphenyls (PCBs) receive a weighting of 15.0, whereas sewage effluent can in general be weighted anywhere between 1.0–3.0 depending on specific content. If discharges are made to groundwater they receive a weighting of 0.5, while surface waters are rated 1.0, and estuaries 1.5.

In the 1991 charging scheme (NRA, 1991c), the financial factor was set at a level to cover the costs of monitoring and enforcement, resulting in a weighting of £202.50. This implies that a "standard" discharge of "storm discharges at sewage treatment works", into surface waters (rivers), up to $100\,m^3$/day, would receive a charge of £202.50/year, while the maximum that could be levied would be £63 787.50. In fact, the cost-recovery nature of the charging regime results in charge levels which bear little necessary relation to the damage caused by the discharge. For instance, the levy on discharges to groundwater of 0.5 is because the NRA does not attempt to monitor discharges of effluent to groundwater, because of the high cost of so doing, despite the acknowledged greater severity, and potential irreversibility, of groundwater pollution (Royal Commission on Environmental Pollution, 1992).

Moreover, consents are allocated on the basis of the expected impact of the licensed effluent in relation to the achievement of the SWQO for the receiving water in question. Consent is granted so long as the effluent would not prevent the SWQO from being achieved. No consideration is given to the value of the consent to the licensee, i.e. to the costs of abatement. Similarly, consents cannot be traded between dischargers in response, for instance, to changes in market condition, improvements in technology, or even simple disparities between dischargers' treatment costs. The result is that the SWQO for each water, and the sustainability objective generally, are likely to be met at a cost which is significantly greater than necessary, because pollution abatement will not necessarily be carried out by those who find it least costly to do so.

A glance at the simple economic model of pollution control is sufficient to demonstrate this (e.g. Dubourg, 1994; Hanley, 1993). This model holds that, for a given abatement cost level, more pollution control should be targeted at the more serious pollutants. Similarly, for a given pollutant, more control should be targeted at those sources which finds it cheaper to abate. In other words, pollution control targets should be flexible across polluters, and should take account of the severity of their pollution, and how cheaply they can abate. This will ensure that the *most* environmental improvement is achieved at the *least* cost of economic resources.

The current system in England and Wales does set effluent charges higher for discharges containing more noxious chemicals, but then encourages more damaging discharges to groundwater by giving them less weight in the charging scheme. Moreover, no consideration of relative abatement costs enters in the decision to license discharges. As a result, it is likely that those who abate their pollution under the current system are not necessarily those who would find it cheapest to abate.

The resource cost savings gained through implementing the efficient, flexible solution as compared with more uniform regulation can be significant. as illustrated by a study by Hanley and Moffatt (1993) of pollution control in the Forth Estuary in Scotland. (Although the regulatory bodies governing water quality management in Scotland differ from those in England and Wales, the pattern of regulation is sufficiently similar to make the study relevant for our purposes.) Although relatively clean compared with similar water bodies, the Forth Estuary faced problems of low dissolved oxygen levels (< 4 mg/l), over a considerable part of its length, during periods of warm weather and low flow. Moreover, the Forth River Purification Board's aim was gradually to improve environmental quality standards in the estuary. Accordingly, Hanley and Moffatt (1993) constructed a linear programming model of pollution control in the Forth, incorporating knowledge of the abatement technology available to all major relevant polluters, both industrial and public treatment works, combined with a model of physical water quality.

They found that, when compared with cutbacks in BOD loading which were uniform across polluters, the least-cost, flexible solution to achieving a range of BOD reduction targets entailed between 9.3 and 27.9% of the resource costs, depending on the target and the time period over which it was to be achieved. In other words, uniform cutbacks were between four and ten times more expensive than the efficient solution. For the case of the dissolved oxygen "sag", meanwhile, the resource savings of the least-cost solution compared with *flexible* regulation, concentrating on large reductions by only those two sources primarily responsible in this case, amounted to only 14%, or £166 000 per year. Clearly, in this simple case, uniform regulation could quite feasibly be tailored to suit actual economic and environmental conditions, implying that a sophisticated regime of pollution taxes or tradeable discharge permits, admittedly by no means simple to implement (Hahn, 1989), might not be necessary to achieve considerable improvements over a uniform regulation solution.

Concluding remarks

A meaningful definition of sustainable water management requires that we are able to define freshwater as a form of critical capital. At least in countries with plentiful water supplies and high levels of non-essential use, this might not be realistic. Sustainable water management might then end up meaning little more than full-cost pricing. However, freshwater can still have an important role to play within a general sustainability objective. For such a general objective requires, at least (according to the constant capital model, and subject to some caveats) that the *rents* of the depletion of a resource should be reinvested in available substitutes (Hartwick, 1977). Hence, although we might be prepared to allow unsustainable water use in terms of the simple rules presented above, we should still like to ensure that the rents from that "unsustainability" are reinvested elsewhere.

Indeed, a critical capital assessment of water might still be justified on the following grounds. That is, that the compensating investments which are envisaged in the "constant capital" model of sustainable development presuppose a particular institutional set-up which might be regarded as somewhat unrealistic in practice. Hence, we might be unable to guarantee sufficient investment elsewhere in the economy to compensate for disinvestment in the water sector, even if capital-substitution opportunities in the economy are extensive. An interventionist, sectoral policy might then be viewed as a "second best" sustainability policy in response to the absence of the institutional arrangements adequate to produce a sustainable outcome without such intervention.

Nevertheless, an assessment of the likely substitution possibilities in the water sector suggests that a "first best" definition of sustainability would allow considerable flexibility in the way water resources are managed in practice. Sustainability is a purposely long-run allocation criterion based primarily on a concern for the well-being of future generations. As a result, declines in environmental quality, whether in the water sector or elsewhere, should not be seen as necessarily implying non-sustainability. The relevant question is whether the environmental damage is the result of a structural overutilization of resources—i.e. a fundamental misallocation—or of natural processes of economic development and structural change. Indeed, to increase pollution in the early stages of development may well represent a justifiable social decision for a developing economy, until at some "mature" point, attitudes to the environment change and the economy becomes oriented towards more environmental objectives (Winpenny, 1995). So long as savings and investment rates are sufficient to offset any capital depreciation (admittedly no simple requirement), then development can be sustainable, and any environmental damage can be regarded as simply representing a change in the economy's capital mix.

It would seem, then, that sustainability and sustainable water management might not represent particularly stringent criteria for the allocation of environmental resources. They constrain economic behaviour over the longer term, and might prevent certain courses of action on grounds of precaution in the face of scientific and other uncertainties (Pearce *et al.*, 1994), but they say little about how resources should be allocated in current periods. This merely emphasizes the role that standard economic analysis continues to have even within a sustainability framework. For sustainability, environmental resources still needed to be valued according to the standard economic approaches. This is admittedly by no means an easy task, but progress is being made, to the extent that estimates of economic value gained through the application of the survey-based contingent valuation method are now acceptable as evidence in U.S. damages cases. This even applies to the estimation of *existence value*, generally regarded as the most taxing class of economic value to estimate (Dubourg *et al.*, 1997). Even given the sometimes high cost of conducting valuation studies, the evidence we have presented suggests that this will be dwarfed by the savings to be gained from improving existing water resource allocations. The technique of benefits transfer promises to reduce greatly the costs of arriving at environmental values (Bateman *et al.*, 1995). It is this framework, not sustainability, which is relevant for the great majority of environmental damage cases in the water sector and elsewhere.

The development of sustainability concepts and policies is welcome, as it serves to emphasize the importance of environmental issues, as well as the strong ethical component of many environmental questions. However, sustainability should not be seen as a panacea for environmental problems, and policy-makers should avoid the temptation to place what are basically very long-run considerations before more immediate, and quite probably more pressing. concerns. Indeed, the "open arms" with which sustainability concepts have been embraced in certain policy-making circles contrasts strongly with the reluctance to address such issues as pollution control and environmental tax reform. This reluctance not only greatly increases costs now, but also seriously reduces the probability that sustainability objectives will even be achieved in the first place.

Ultimately, the message needs to be understood that sustainable development and environmental economic analysis are not substitutes but important complements to each other. Perhaps then it might be realized that placing water planning on a more economic footing could well produce the classic "double dividend": Not only would it facilitate the achievement of sustainability in the water sector, it would also greatly improve the efficiency of current and future water allocation decisions

Acknowledgements

The financial support of the National Rivers Authority, the Economic and Social Research Council (ESRC) (Fellowship number W 103 06 1001), WRc plc, and the Commission of the European Communities is gratefully acknowledged. All views expressed are the author's own. Thanks are also due to Jennifer Barker. CSERGE is a designated research centre of the ESRC.

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